BULLETIN
OF THE
TEXAS ORNITHOLOGICAL SOCIETY

Vol. 49 No. 1–2 December 2016

Published by the
Texas Ornithological Society
www.texasbirds.org

THE TEXAS ORNITHOLOGICAL SOCIETY

2016 OFFICERS

MEMBERSHIP

Any person interested in the birds of Texas may become a member of the Texas Ornithological Society. Individual membership rates are $25.00 per year, $50.00 at the Sustaining level, $15.00 Student, Family $35.00, $400.00 Life, Organizational $25.00, and $500.00 Life w/Spouse. These memberships include print versions of ***Texas Birds Annual***, and the ***Bulletin of the Texas Ornithological Society***. To become a member send check or money order in U.S. funds with complete name and address to: Texas Ornithological Society Memberships, PMB #189, 12468 Austin Highway, San Antonio, Texas 78209. Memberships can also be obtained by consulting www.Texasbirds.org

PUBLICATIONS

The ***Bulletin of the Texas Ornithological Society*** is an annual[1] journal devoted to the biology and conservation of the birds that occur in Texas and adjacent areas.

Texas Birds Annual is published each year and includes new birding areas, recent media, vagrants and noteworthy birds from the previous year, articles on bird conservation issues and species profiles..

TOS News is a tri-annual full color electronic newsletter produced by TOS. The newsletter is no longer a membership benefit but "open access" on our webpage www.texasbirds.org.

Occasional Publication Series is published on an irregular basis on scientific works too long for publication in the *Bulletin*. When published it is provided free to all members as an extra benefit of membership during the year of publication.

Submission to all publications should be e-mailed to: jclintoneitniear@gmail.com

Changes of address and queries about memberships or missing copies: E-mail TOSmember@yahoo.com or submit in writing to the address above.

Additional copies of publications can be obtained by contacting the editor.

PERMISSION TO COPY

The Texas Ornithological Society hereby grants permission to copy articles (in whole or in part) appearing in ***The Bulletin of the Texas Ornithological Society (Bulletin) and Texas Birds Annual (TBA)*** for personal use, or educational and/or personal use, without payment, provided the material reproduced includes the statement "©2011 The Texas Ornithological Society" and contains the full citation, including names of all authors. TOS does not maintain copyright for photographic material therefore does not allow any reproduction of such material without written consent of the photographer. Any use not specifically granted here, and any use of Bulletin or TBA articles or portions thereof for advertising, republication, or commercial uses, requires prior approval from the Editor (jclintoneitniear@gmail.com).

---

[1] Starting with Vol. 42 both issues were combined..

Cover Photo: Immature Red-tailed Hawk *Buteo jamaicensis*, Photo by Greg Lasley. Balmorhea Lake, Reeves Co., Texas, 16 December 2013.

Frontispiece. Northern Bobwhite (*Colinus virginianus*). Art compliments of Lynn Delvin..

BULLETIN OF THE

TEXAS ORNITHOLOGICAL SOCIETY

LAND BIRD ABUNDANCE AND BRUSH REDUCTION IN THE TEXAS COASTAL PRAIRIE

Eloy Alejandro Lozano-Cavazos[1,3,4], Leonard A. Brennan[1], William P. Kuvlesky, Jr.[1]
Fidel Hernández[1,] and Wade C. Harrell[2]

[1]*Caesar Kleberg Wildlife Research Institute, Texas A&M University – Kingsville, 78365*
[2]*U.S. Fish and Wildlife Service Aransas National Wildlife Refuge, Austwell, TX 77950*

ABSTRACT.—Reversing the ongoing decline of grassland birds in North America will require effective and sustainable management strategies for restoring their habitats and populations. Traditional management methods such as roller-chopping and prescribed fire may be an option to restore and maintain habitats for grassland birds in mesic environments where brush has increased in the past 100 years such as the coastal prairies in southern Texas. We evaluated estimates of breeding and wintering land bird population abundance in relation to two approaches to brush reduction: a combined treatment based on the successive use of roller-chopping, followed with prescribed fire, and the application of herbicides to individual shrubs (combined treatment); and an application prescribed summer fire (summer fire treatment). These two treatments were compared between each other and against plots sampled on open grassland areas which served as comparative controls that represented open grassland vegetation with little or no brush cover. Land bird species richness on the combined and summer fire treatments was consistently greater than species richness on the control sites during the three winter and two summer seasons of the study. Land bird species richness decreased on both treatments during winter and summer seasons over the course of the study. Overall relative abundance of individual species was greater on both the combined and the summer fire treatment sites compared to the control sites. A rapid recovery of dense, herbaceous vegetation in response to above-average rainfall was apparently an influential factor that corresponded to sharp decreases in the abundance of many land bird species over the three years of the study. A range of between 10% and 30% of brush canopy cover scattered throughout the landscape should be considered for restoring vegetation in order to maximize bird species richness in a coastal prairie setting. Once any kind of treatment is applied to reduce brush cover for grassland bird management, further maintenance should be considered to maintain presence and heterogeneity of herbaceous vegetation.

During the mid-nineteenth century Europeans eliminated herds of free roaming wildlife and restricted cattle movements as they settled Texas rangelands. As settlements expanded, continuous grazing of cattle along with fire suppression promoted brush encroachment that changed grasslands into the woody plant communities that dominate rangelands today (Hamilton et al. 2004). Changes of open grassland and savannas to more woody landscapes have transformed the

---

[3]E-mail: alejandro.lozano-c@uaaan.mx

[4]Current address: Departamento de Recursos Naturales Renovables Universidad Autonoma Agraria Antonio Narro Area de Fauna Silvestre, Saltillo, Coahuila, Mexico.

avian community composition from a grassland to a shrubland bird community. For instance, the diversity and abundance of many species that are grassland bird specialists have declined whereas bird species that have affinities to shrub habitats have increased (Askins 2000).

The estimated decline of tall grass, short grass, and mixed grass prairie in North America has been about 90, 80, and 30% respectively (Samson and Knopf 1994, Knopf 1994). Additionally fragmentation and rangeland deterioration have exacerbated the widespread loss of grassland habitat in North America (Brennan and Kuvlesky 2005). Moreover, Northern Bobwhite (All scientific names can be found in Appendix 1) as a part of the grassland bird guild has declined over the last four past decades throughout its range (Peterson et al. 2002).

The overall goal of this study was to evaluate the effectiveness of two habitat management approaches as techniques used to reduce brush cover on large tracts of the Coastal Prairie Ecoregion in South Texas. The 2 management practices evaluated included (1) a combined treatment based on a mechanical (roller chopper) followed with a prescribed fire and then an herbicide application, and (2) summer prescribed fire treatment alone. Open grasslands were used as control-comparison sites. Specifically, the objectives were to: (1) document avian species richness during two breeding and three wintering seasons across the treatments and control sites; and (2) estimate avian abundance for individual species during two breeding and three wintering seasons across the treatment and control sites, and (3) quantify changes in vegetation on the treatment and control sites. We hypothesized that the combined (roller chopped-fire-herbicide), and summer prescribed fire treatment sites would have greater species richness and greater overall land bird abundance than the control sites because of the presence of remnant woody vegetation on the treatment sites.

METHODS

Study Area

The study area was on the 20,234 hectare Vidaurri Ranch, 24 kilometers north of Refugio, Texas, in Refugio and Goliad counties. It was dominated by brush species such as huizache and mesquite ranging from <1% to >60% brush canopy. Soil types are fine sandy loams and Edroy clays soils (USDA-NRCS, 2008). The climate was humid subtropical with hot summers (Hatch et al. 1999). The mean annual precipitation was 762 mm, with a summer average of 463 mm and a winter average of 299 mm. The daily mean temperature was 22 (°C) with a mean daily maximum of 28 (°C) and mean daily minimum of 16 (°C) (Norwine and John 2007). The combined treatment, summer fire treatment and no-treatment control plots were located on Vidaurri Ranch; an additional study area representing a no-treatment control plot was located at about 30 km south of Vidaurri Ranch at the Aransas National Wildlife Refuge owned by United States Fish and Wildlife Service (USFWS) in Aransas County, Texas. The Aransas control plot was located on the Tatton management unit and was dominated by open grasslands with scattered brush (approximately 5%) maintained by a multi-patch burning regime across its 2,318 hectares. Additional study area details can be found in Lozano-Cavasos (2009).

Experimental Design

This project was designed to evaluate how land bird abundance in the Texas coastal prairies would vary in relation to brush reduction by comparing three treatment categories: (1) a 3-year combined brush management treatment that used roller-chopping, prescribed fire and herbicide applications; (2) a single year application summer prescribed fire, and (3) a "control" site that represented the desired set of target conditions (extensive grassland with little or no brush cover) sought through restoration.

The 3-year combined treatment consisted of mechanical woody plant reduction via a roller-chopper pulled by D-8 bulldozer during the initial year of the treatment. During the following year a summer prescribed fire was used to further reduce woody plants. The final year of the treatment consisted of individually treating remaining brush with soil-applied hexazinon herbicide (Velpar L©). Three different blocks (south, middle, and north) of about 200 ha each on the combined treatment were treated in different years with a total size of 600 ha. The combined treatments applied on south block began in 2002 and ended in 2004, on the middle began in 2003 and ended in 2005, and on the north block began in 2004 and ended in 2006. Thus, when our sampling began in 2006 the combined treatment plots were between zero and two years post-treatment. On the summer fire treatment, a prescribed burn was applied during summer (June) 2004 on a single 600 ha block. The Vidaurri control site was burned during the winter of 2003. It had less than 1% of brush canopy cover on its 200 hectares. Four sub-management units encompassed the Aransas control site (T10,T11,T12, and T15) with a total 737 hectares. By 2007, when Aransas control was evaluated, the last prescribed burn in the T10 subunit was in 2004, in T11 subunit 2006, in the T12 subunit 2005, and in T15 subunit 2004. Thus, prescribed burns on previous subunits were between 1 and 3 years post treatment when the study began.

A restricted randomized study design (Morrison et al. 2001) was used to quantify the relationships of the treatment regime(s) on the land bird community. Therefore, 2 treatments and 1 control were randomly assigned to seven 200-ha experimental units, three corresponded to combined treatment, three corresponded to summer fire treatment, and one corresponded to Vidaurri control site. The Aransas control was surveyed during the 2007 breeding season.

Breeding Season Bird Point Counts

The sampling units for breeding season birds were point counts, and the placement and location of each point count depended on the area and/or the shape of each treatment. For instance, 30 point counts 400 meters apart were placed throughout the treatments during the first breeding season 2006. Eleven points corresponded to the combined treatment, 14 points corresponded to the summer fire treatment, and 5 points corresponded to Vidaurri control. In order to obtain more complete coverage, during the second breeding season (2007) the number of points increased from 30 to 51 points and were placed 200 meters apart throughout the treatments. Twenty one point counts corresponded to the combined and summer fire treatments respectively, whereas 9 point counts corresponded to the Vidaurri control. During breeding season 2007, the Aransas control site was added in order to replicate the representation of open grasslands in the study. The Aransas control had 21 point counts.

The breeding bird community was sampled during two consecutive breeding seasons (summer 2006 and 2007). Monitoring was conducted during 3 periods between mid-May and July. Point counts were used to monitor bird species diversity and abundance (Hutto et al. 1986, Gutzwiller 1991, Hamel et al. 1996). All point counts were geo-referenced using Garmin E-trex global positioning system (GPS) and the center of each point count was marked with a colored stake. The GPS unit was used to re-locate each point count station. Individual visual bird observations were truncated to a100-meter radius to make abundance comparisons between seasons; observations beyond the predetermined radius (100 m) were added to estimate species richness per treatment. Surveys began at sunrise and continued for 4 hours to minimize hourly variation in detection rates. Visual and auditory observations were recorded for 8 minutes within each point count station after a 2 minute pause prior to arriving at each point count. Species that were present during the two consecutive breeding seasons within any treatment and had at least 1 individual detected per point count were subjected to analysis of means and 95% confidence intervals between years. Species that did not meet the previous criteria were considered rare, only the mean number of individuals detected per point count for these species were reported.

Winter Season Bird Count Transects

Because detections of wintering birds are more often based on sightings than calls, the sampling units for the wintering bird counts were transects. The placement and location of each transect depended on the area and/or the shape of each treatment. For example, 14 transects of variable length at least 500 meters apart were placed throughout the treatments. Six transects corresponded to the combined treatment with a total length of 5,250 meters, six transects corresponded to the summer fire treatment with a total length of 6,750 meters, and two transects corresponded to the Vidaurri control with a total length of 3,000 meters in Vidaurri Ranch; 2 additional transects of 1,500 meters were set up in Aransas control with a total length of 3,000 meters during the third and last winter season (2007-2008).

The land bird community was sampled during three consecutive winter seasons (2005-2006, 2006-2007, and 2007-2008). Sampling was conducted during 3 periods between mid November and February. Line transects were used to monitor bird species diversity and abundance. All transects were geo-referenced using Garmin E-trex global positioning system (GPS) and the beginning and end of each transect was marked with a colored stake. The GPS unit was used to re-locate each transect during sampling. An index of population size was created by dividing the number of individuals of each species seen per kilometer of transect. Species that were present during at least two consecutive winter seasons within any treatment and had at least 1 individual detected per kilometer were considered abundant and thus were subjected to analysis of means and 95% confidence intervals across years. Species that did not meet the previous criteria were considered rare, only the mean number of individuals detected per kilometer for these species were reported.

Vegetation Structure and Composition

We evaluated vegetation structure on each of the treatment and control sites. During the breeding season we used the 21 point count locations as centers of sampling plots to evaluate vegetation structure on combined treatment, summer fire treatment, and the Aransas control, whereas 9 point counts were used on the Vidaurri control. The size of the plots was 0.19 hectares. Plots were 200 meters apart, and each one contained 4-25 m line intercepts, by placing the first line intercept at random cardinal direction (e.g. N, S, E, W) and then the subsequent line intercepts in a clockwise direction. Line intercepts served to estimate brush canopy cover and density per treatment. Total brush density was estimated by counting the number of brush plants one meter of width to the right of the line transect. Brush density was estimated by three height classes (0-1, 1-2, and > 2 meters)

For the winter season vegetation sampling on permanent plots were used along the same transects that we used for surveys of grassland birds. Plots of 0.19 hectares in size were set up at the beginning of each transect and then every 150 meters, hence, the number of plots per transect depended upon the length of each transect. The combined treatment had a total of 41 plots, the summer fire treatment had 51 plots, and Vidaurri control had a total of 22 plots.

Each plot contained 3-25m line intercepts (Canfield 1941) that were used to estimate brush canopy cover by placing the first line intercept at a random cardinal direction (e.g. N, S, E, W) and then the subsequent line intercepts in a clockwise direction. For example, the first reading was at east cardinal direction and then the second one was on south cardinal direction, and so on. Each line intercept was in a radial direction from the center of each plot. Brush density was estimated by using each line intercept for counting the number of individual species inside of 1 meter width to the right of the transect. Herbaceous vegetation was evaluated using Daubenmire frames (Daubenmire 1959) along each line transect. The combined treatment had a total of 369 frames, the summer fire treatment had 459 frames, and Vidaurri control had a total of 198 frames during the first winter season 2005-2006. During the second winter season 2006-2007 Daubenmire frames were increased to a total of 492 on the combined treatment, 612 on the summer fire treatment, and 264 on the Vidaurri control.

In order to evaluate the grass, forbs, litter and bare ground cover 3 Daubenmire frames (25 x 50 cm) were set up along each line transect by placing the first one at 9 meters from the beginning of the transect and two subsequent frames every 8 meters. In order to avoid disturbance on Daubenmire samples we set up the frames in a perpendicular direction from the line transect 2 meters away. Visual obstruction was estimated by using a 2-meter pole segmented by 0.5 m intervals and situated at the center of the plot. The readings were taken 15 meters away from the center on the plot using four cardinal directions. Grass height was measured on the right corner of each Daubenmire frame.

Data Analysis

Bird species richness was compared across treatments and control sites. Sampling effort curves were used to determine the maximum number of species present across treatments and control sites. Sampling effort curves also served as a check to make sure that adequate sampling effort was directed at each treatment and control site. Species abundance (based on number of individuals of each species detected per point count or transect) was computed and reported as means with associated 95% confidence intervals (Johnson 1999). Vegetation structure comparisons were also reported as means with 95% confidence intervals. All calculations were performed using STATISTICA software, Version 8.0.

RESULTS

Breeding Birds

Species Richness.—Twenty-seven species of birds were detected during the two breeding seasons of this study (Table 1, Appendix 2). Bird species richness was consistently greater on combined treatment and summer fire treatment sites than it was on the control sites (Figure 1). Species richness on the summer fire and combined treatments were about twice as great (1.9 and 1.7 times respectively) than on the Vidaurri control. During the second breeding season (2007), the combined treatment had 12 species and the summer fire treatment had 10 species. Species richness on the combined treatment and summer fire treatment were 3.0 and 2.5 times greater than species richness on the Vidaurri control, respectively. Species richness on the combined treatment and summer fire treatment were 1.5 and 1.3 times greater than on Aransas control.

Table 1. List of grassland bird species found on combined treatment [CT], summer fire treatment [SF], Vidaurri control [VC], and on Aransas control [AC] during two consecutive breeding (May 15 to July 31) seasons 2006 (1) and 2007 (2), Goliad Prairie, Texas.

Species	Treatments						
	CT		SF		VC		AC
	1	2	1	2	1	2	2
Ash-throated Flycatcher				X			
Blue Grosbeak	X		X				
Brown-crested Flycatcher			X	X			
Brown-headed Cowbird	X		X	X			
Cassin’s Sparrow	X		X		X		
Carolina Chickadee		X	X				
Cattle Egret					X		
Cave Swallow			X				
Clay-colored Robin		X					
Cliff Swallow	X	X	X		X		X
Common Nighthawk	X						
Dickcissel		X		X		X	X
Eastern Meadowlark	X	X			X	X	X
Golden-fronted Woodpecker			X				
Ladder-backed Woodpecker	X		X	X			
Lark Sparrow			X		X		
Mourning Dove	X	X	X	X	X	X	X
Northern Bobwhite	X	X			X		X
Northern Cardinal	X	X	X	X			X
Northern Mockingbird	X	X	X	X	X		X
Painted Bunting	X		X				
Red-winged Blackbird	X						
Scissor-tailed Flycatcher	X	X	X	X	X	X	X
Vermilion Flycatcher		X					
White-tailed Hawk			X				
Yellow-billed Cuckoo	X		X	X			
TOTAL	15	12	17	10	9	4	8

Overall species richness within treatments decreased as time progressed. Species richness decreased from 15 to 12 species on the combined treatment, from 17 to 10 species on the summer fire treatment, and from 8 species to 4 on the Vidaurri control. The Aransas control had twice as many species compared to the Vidaurri control during

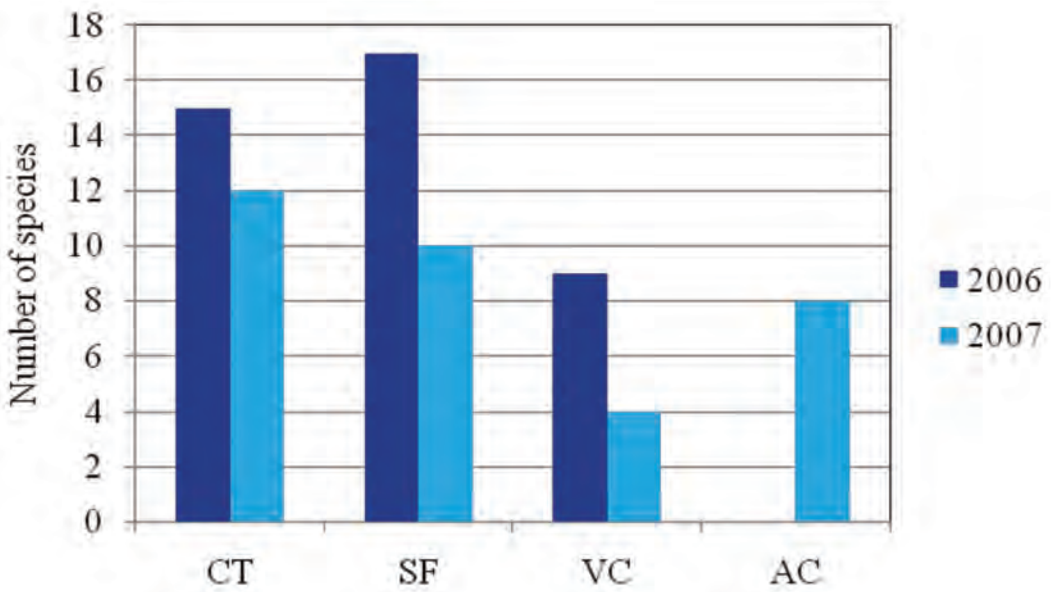


Figure 1. Grassland bird species richness across combined treatment [CT], summer fire treatment [SF], Vidaurri control [VC] and Aransas control [AC] during two consecutive breeding (May 15 to June 31) seasons 2006 and 2007.

the second breeding season. Even though species richness both treatments decreased in during the second breeding season, the number of species was still greater on both treatments compared to the control sites during the two consecutive breeding seasons.

Sampling Effort Curves.—Approximately 22 point counts were needed to determine the number of species present on the combined treatment and Vidaurri control. For the summer fire treatment 12 point counts were needed, and during the second breeding season 28 point counts were needed to estimate species richness on the Aransas control site (Figure 2).

Abundance by Treatments and Years.—The 3 most abundant breeding bird species, which ranged from 0.04 detections per point to 3.1 detections per point were Cliff Swallow, Eastern Meadowlark, and Northern Mockingbird. These 3 species accounted for 58% of the total number of detections across all of the treatments and control sites. The other 24 species, which accounted for 42% of the total detections, ranged from 0.01 per point to 0.5 detections per point (Appendix 2). Of these 24 species, abundance by treatment and season were highly variable.

Northern Mockingbird abundance on the combined treatment was statistically similar between the first and second breeding seasons. Even though 95% confidence intervals overlapped, the mean abundance decreased 31% from the first breeding season (mean ± 95% C.I.; 1.6 ± 1.2 individuals/km) to the second (0.5 ± 0.2 individuals/km). Northern Mockingbird abundance on the summer fire treatment exhibited a similar trend. Abundance was similar between the first (1.3 ± 0.9 individuals/km) and second breeding season (0.2 ± 0.2 individuals/km); however, abundance decreased by 15% . Northern Mockingbird on the Vidaurri control was present only during the first (16.4 ± 11.8 individuals/km) breeding season, and then its abundance declined to zero. Northern Mockingbird abundance decreased on both treatments and Vidaurri control. On the combined treatment, summer fire treatment, and on the Vidaurri control there was a decrease of 69%, 85%, and 100%, respectively (Figure 3).

Cliff Swallow abundance on the combined treatment was similar between the first and second breeding seasons. Even though 95% confidence intervals overlapped, the mean abundance increased 355% from the first breeding season (mean ± 95% C.I.; 0.9 ± 0.7 individuals/km) to the second (3.2 ± 1.7 individuals/km). . Cliff Swallow was present only during the first breeding season on the summer fire treatment and Vidaurri control with

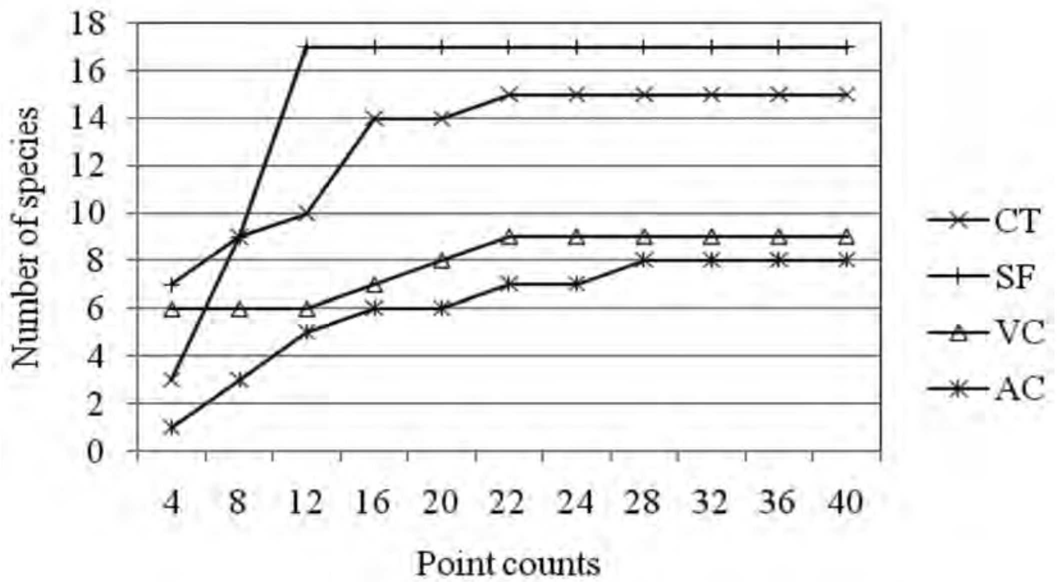


Figure 2. Sampling effort curve for combined treatment [CT], summer fire treatment [SF], Vidaurri control [VC] during breeding (May 15 to July 31) season 2006, and on Aransas control [AC] during breeding (May 15 to July 31) season 2007.

an abundance of (0.1 ± 0.2 individuals/km), and (1.1 ± 0.8 individuals/km), respectively; then its abundance declined 100% by the second breeding season. Cliff Swallow abundance on Aransas control during the second breeding season was (0.8 ± 0.6 individuals/km).

Cliff Swallows exhibited an increasing trend on the combined treatment, and a decrease trend on the

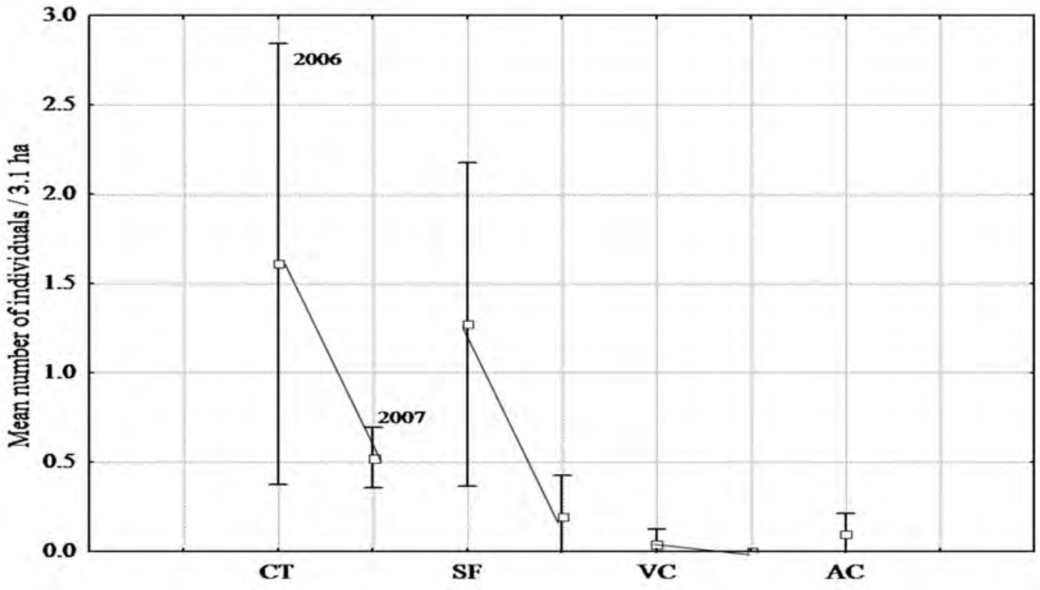


Figure 3. Abundance of Northern Mockingbird (mean number of individuals / 3.1 ha) (Mean ± 0.95 C.I.) on combined treatment [CT], summer fire treatment [SF], and Vidaurri control [VC] during two consecutive breeding (May 15 to July 31) seasons 2006 and 2007, and on Aransas control [AC] during breeding (May 15 to July 31) season 2007.

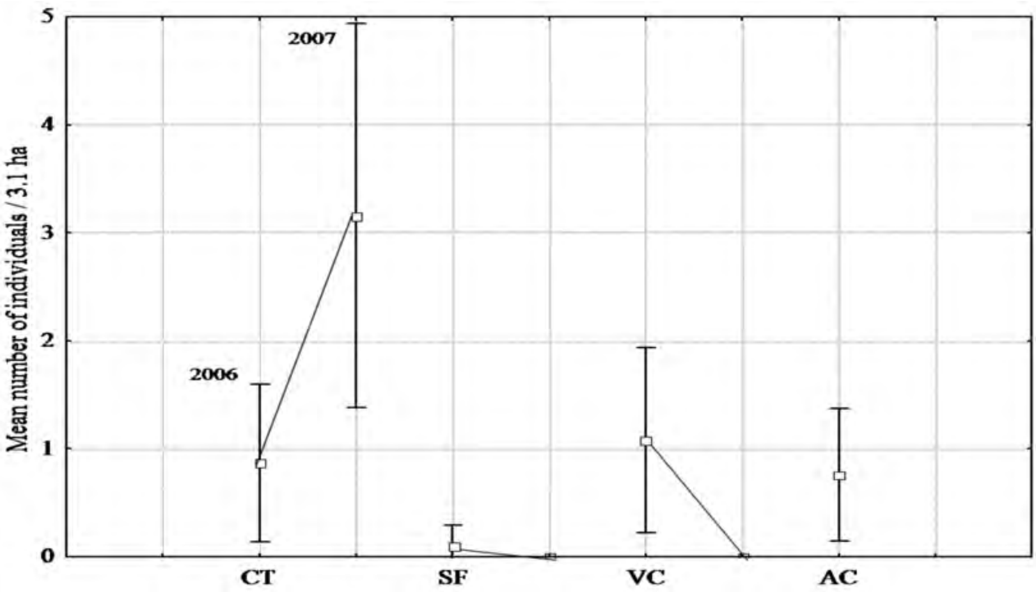


Figure 4. Abundance of Cliff Swallow (mean number of individuals / 3.1 ha) (Mean ± 0.95 C.I.) on combined treatment [CT], summer fire treatment [SF],and Vidaurri control [VC] during two consecutive breeding (May 15 to July 31) seasons 2006 and 2007, and on Aransas control [AC] during breeding (May 15 to July) season 2007.

summer fire treatment and Vidaurri control on the second breeding season. The general pattern that emerged was that abundance was similar between the two breeding seasons. On the combined treatment there was an increase of 355% during the second breeding season, and a nearly100% decrease on the summer fire treatment and the Vidaurri control (Figure 3.4).

Eastern Meadowlark abundance on the combined treatment was statistically similar between the first and second breeding seasons. Even though 95% confidence intervals overlapped, the mean abundance increased 100% from the first breeding season (mean ± 95% C.I.; 0.1 ± 0.2 individuals/km) to the second (0.2 ± 0.1 individuals/km). Eastern meadowlark abundance was similar between the first and second breeding seasons on Vidaurri control. Even though 95% confidence intervals overlapped, the mean abundance decreased 57% from the first breeding season (mean ± 95% C.I.; 1.4 ± 0.5 individuals/km) to the second (0.8 ± 0.4 individuals/km) (Figure 5). Abundance of Eastern Meadowlark on Aransas control during the second breeding season was (0.5 ± 0.2 individuals/km).

Eastern Meadowlark abundance showed an increasing trend on the combined treatment and a decrease on the Vidaurri control. The general pattern was that abundance was similar between the two breeding seasons. On the combined treatment there was an increase of 100% whereas on the Vidaurri control had decreased 43% during the second breeding season however, its abundance was much still greater in Vidaurri control compared with the combined treatment (Figure 5).

Wintering Birds

Thirty-six species of birds were detected during the three consecutive winter seasons in our study (Table 2, Appendix 3). Species richness was consistently greater on the combined treatment and summer fire treatment sites compared to the control sites (Figure 6) during three consecutive winter seasons. During the first winter season (2005-2006), the summer fire treatment had 31 species and was 2.1 times greater than the combined treatment which had 15 species. Species richness on the summer fire and combined treatments were 4.4 and 2.1 times greater than on Vidaurri control site respectively which had 7 species.

During the second winter season (2006-2007), the combined treatment had 25 species and was similar to the summer fire treatment which had 24 species. Species richness on the combined and summer fire treatments were 3.1 and 3 times greater than on the Vidaurri control respectively. During the third winter season (2007-2008), the summer

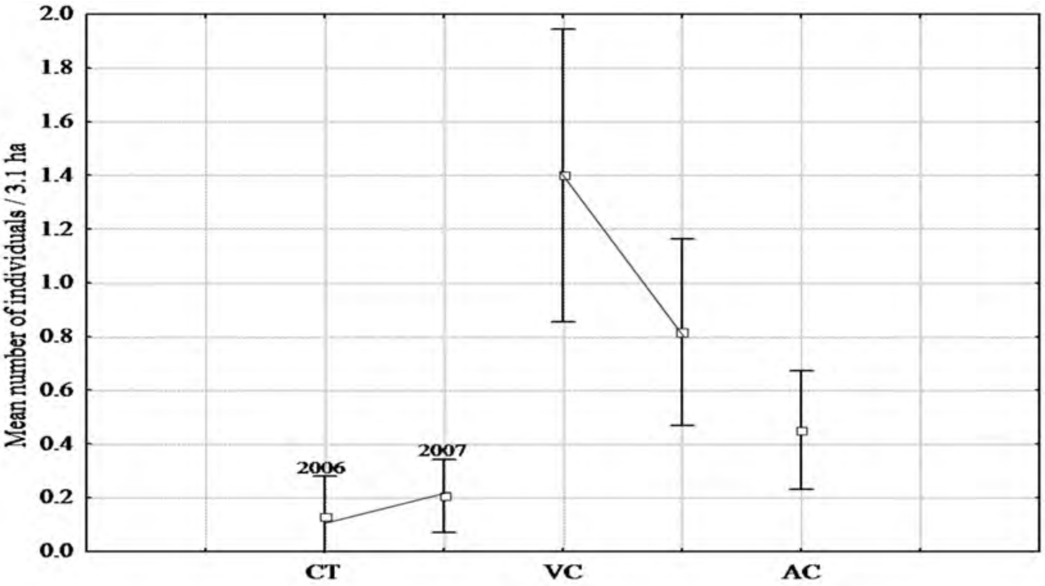


Figure 5. Abundance of Eastern Meadowlark (mean number of individuals / 3.1 ha) (Mean ± 0.95 C.I.) on combined treatment [CT] and Vidaurri control [VC] during two consecutive breeding seasons (May 15 to July 31) 2006 and 2007, and on Aransas control [AC] during breeding (May 15 to July 31) season 2007.

fire treatment had 13 species and again was similar to the combined treatment which had 12 species. Species richness on the summer fire and combined treatment were 2.6 and 2.4 times greater than on Vidaurri and Aransas controls respectively.

Sampling Effort Curves.—Based on the sampling effort to estimate species richness during the first winter season, the summer fire required 18 kilometers to determine the number of species present; for the combined treatment 14 kilometers were needed, for Vidaurri control site 8 kilometers were needed. During third winter on the Aransas control site 10 kilometers were needed (Figure 7).

Abundance by Treatments and Years.—The 9 most abundant wintering bird species, which ranged from 0.05 detections per kilometer to 27.6 detections per kilometer were Eastern Meadowlark, Eastern Phoebe, Grasshopper Sparrow, Mourning Dove, Northern Bobwhite, Savanna Sparrow, Sedge Wren, Sprague's Pipit, and Swamp Sparrow. These 9 species accounted for 86% of the total number of detections across the treatment and control sites. The other 27 species, which accounted for 14% of the total detections, ranged from 0.04 individuals per kilometer to 4.6 individuals per kilometer (Appendix 3).

Savanna Sparrow abundance on the combined treatment was similar between the first and second winter seasons. Even though 95% confidence intervals overlapped, the mean abundance decreased 47% from the first winter season (mean ± 95% C.I.; 25.9 ± 11.5 individuals/km) to the second (12.1 ± 7.7 individuals/km). Savanna Sparrow abundance decreased significantly from its initial abundance by the third winter season (1.5 ± 1.3 individuals/km), a decrease of 94% (Figure 8).

Savanna Sparrow abundance on the summer fire treatment exhibited a similar trend. Abundance was similar between the first (9.4 ± 3.3 individuals/km) and second winter season (7 ± 4.3 individuals/km); however, abundance decreased by 74%. By the third winter season, the abundance of Savanna Sparrow had decreased by 65% from its initial abundance.

Savanna Sparrow abundance on the Vidaurri control decreased from the first (16.4 ± 11.8 individuals/km) to the second winter season (6.9 ± 5.6 individuals/km) and the second to the third winter season (1 ± 1.3 individuals/km), a decrease of 94% (Figure 8).

Eastern Meadowlark abundance on the combined treatment increased 34% from the first winter season (mean ± 95% C.I.; 4.3 ± 3.2 individuals/km) to the second (5.8 ± 2.8 individuals/km). By the third winter season, Eastern Meadowlark abundance had decreased

Table 2. List of grassland bird species found on combined treatment [CT], summer fire

Species	Treatments									
	CT			SF			VC			AC
	1	2	3	1	2	3	1	2	3	3
American Kestrel	X	X	X	X	X				X	
American Pipit				X						
American Robin		X		X	X					
Ash-throated Flycatcher		X		X						
Bewick's Wren				X						
Blue-gray Gnatcatcher		X		X	X	X				
Burrowing Owl		X								
Crested Caracara					X					
Carolina Chickadee				X						
Cassin's Sparrow				X						
Clay-colored Sparrow	X	X		X	X					
Eastern Bluebird				X						
Eastern Meadowlark	X	X	X	X	X	X	X	X	X	X
Eastern Phoebe	X	X	X	X	X	X				
Field Sparrow				X						
Golden-fronted Woodpecker	X	X	X	X	X					
Grasshopper Sparrow	X	X		X	X			X		
House Wren		X		X	X	X				
Inca Dove						X				
LeConte's Sparrow		X		X	X	X				
Lincoln's Sparrow		X		X	X					
Loggerhead Shrike	X	X	X	X	X		X	X	X	X
Mourning Dove	X	X	X	X	X	X	X	X		
Northern Bobwhite	X	X		X	X					
Northern Cardinal	X	X		X	X	X				
Northern Mockingbird	X	X	X	X		X				
Red-shouldered Hawk					X					
Red-tailed Hawk		X		X						
Ruby-crowned Kinglet		X	X	X	X	X				
Savanna Sparrow	X	X	X	X	X	X	X	X	X	X
Sedge Wren	X	X	X	X	X	X	X	X	X	X
Sprague's Pipit	X	X		X			X	X		
Swamp Sparrow		X	X		X	X				
Vesper Sparrow	X	X		X	X					
White-tailed Hawk			X	X	X		X			
White-tailed Kite				X	X			X		X
TOTAL	15	25	12	31	24	13	7	8	5	5

[SF], Vidaurri control [VC], and Aransas control [AC] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006 (1), 2006-2007 (2), and 2007-2008 (3), Goliad Prairie, Texas.

from its initial abundance to 1.4 ± 0.5 individuals/km) a decrease of 67%. Eastern Meadowlark abundance on the summer fire treatment exhibited a similar trend. Abundance was similar between the first (11.9 ± 12.6 individuals/km) and second winter season (3.6 ± 2.3 individuals/km); however, abundance decreased numerically by 30%. By the third winter season, even though 95% confidence intervals overlapped, the Eastern Meadowlark abundance had decreased from its initial abundance by 87% (1.6 ± 2.5 individuals/km). Eastern Meadowlark abundance on the Vidaurri control also was similar during the first (6 ± 4.3 individuals/km) and second winter season (6 ± 4.6 individuals/km). Eastern Meadowlark had decreased significantly by the third winter season (0.9 ± 0.8 individuals/km), a decrease of 85%. Eastern Meadowlark abundance was similar

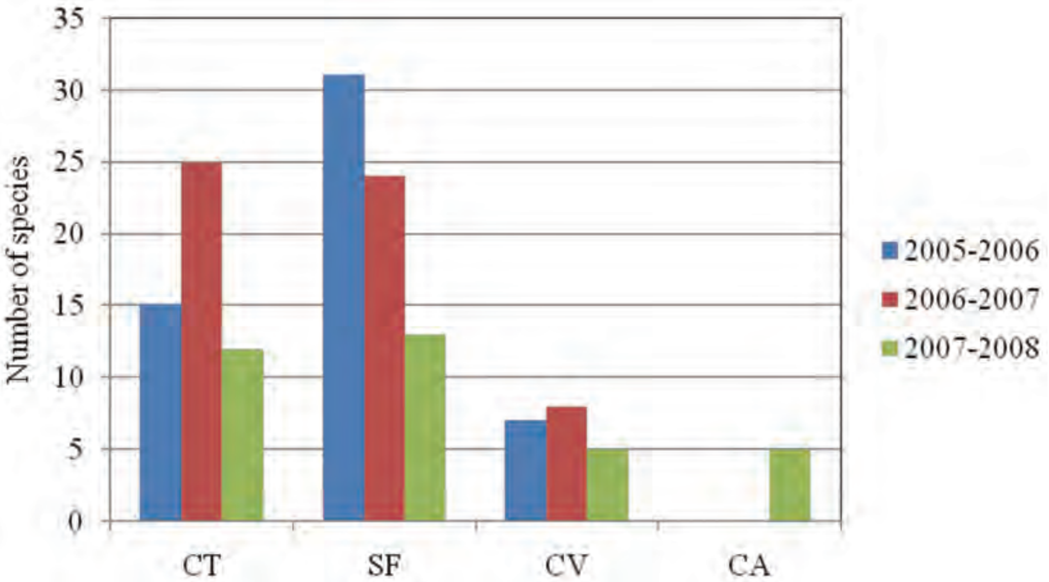


Figure 6. Grassland bird species richness across combined treatment [CT], summer fire treatment [SF], Vidaurri control [CV], and Aransas control [CA] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008.

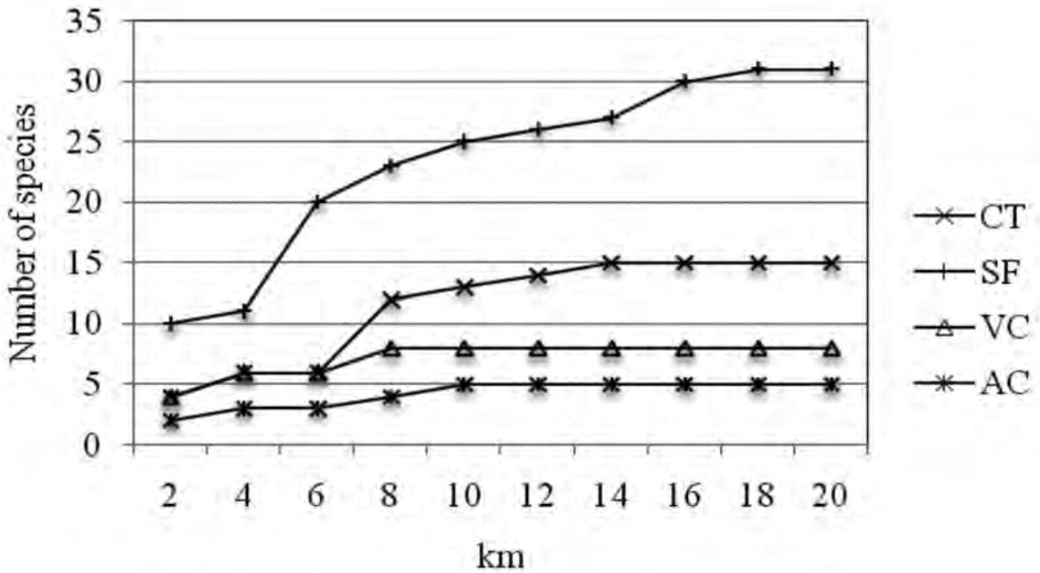


Figure 7. Sampling effort curve for combined treatment [CT], summer fire treatment [SF], and Vidaurri control [VC] during winter (Nov 15 to Feb 28) season 2005-2006, and on Aransas control [AC] during winter (Nov 15 to Feb 28) season 2007-2008. km = length of transects.

between Vidaurri and Aransas controls. Although abundance of eastern meadowlark fluctuated on the combined treatment, all treatments exhibited a general downward trend. Mean abundance declined 67%, 87%, and 85% on the combined treatment, summer fire treatment and Vidaurri control respectively. The abundance of Eastern Meadowlark during the third winter season on Aransas control was similar to that found on the same season on Vidaurri control (Figure 9).

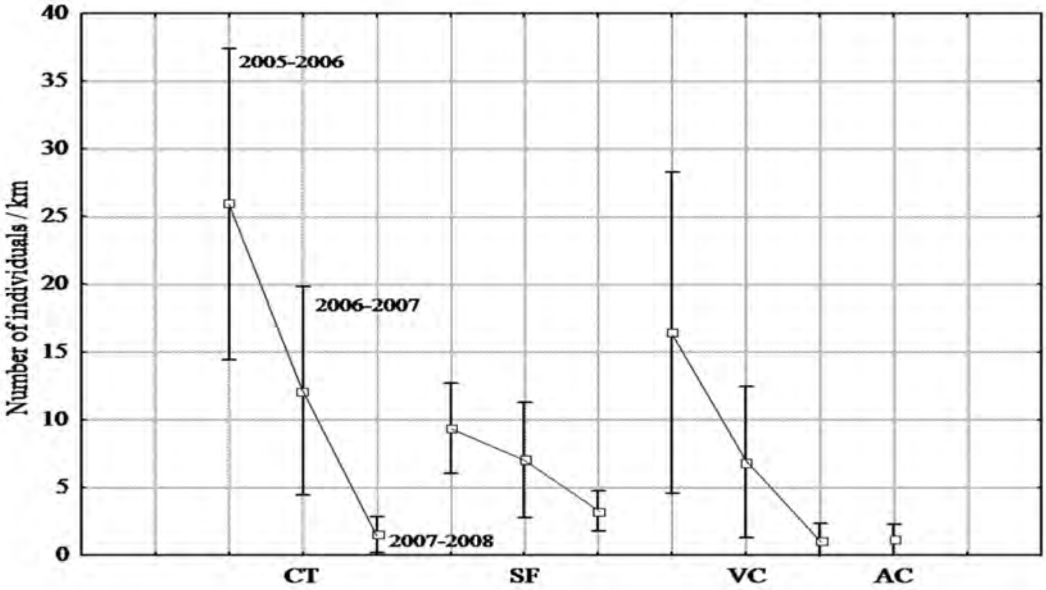


Figure 8. Abundance of Savanna Sparrow (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT], summer fire treatment [SF], and Vidaurri control [VC] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008; and on Aransas control [AC] during winter season (Nov 15 to Feb 28) season 2007-2008.

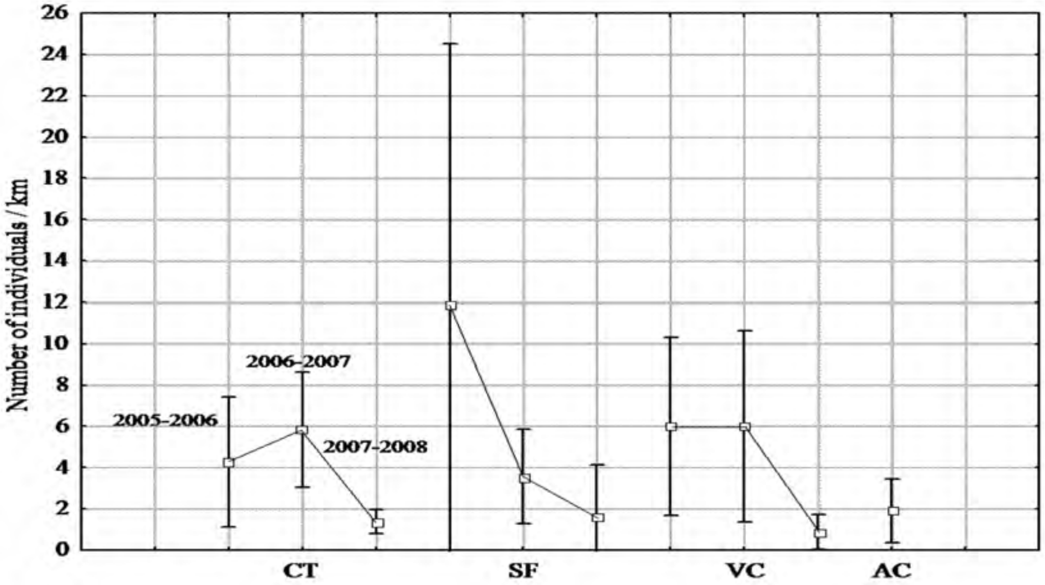


Figure 9. Abundance of Eastern Meadowlark (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT], summer fire treatment [SF], and Vidaurri control [VC] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008; and on Aransas control [AC] during winter season (Nov 15 to Feb 28) season 2007-2008.

Northern Bobwhite abundance decreased on the combined and summer fire treatment plots. Abundance on the combined treatment declined from the first (2 ± 2.3 individuals/km) to the second winter season (1.1 ± 1.7 individuals/km) by 55%. By the end of the third winter season, Northern Bobwhites were absent. Northern Bobwhite abundance on the summer fire treatment declined by 44% between the first (2.5 ± 2.9 individuals/km) and second winter season (1.1 ± 1.4 individuals/km). By the end of the third winter season, Northern Bobwhite were absent on the summer fire treatment; they were never detected on the control sites (Figure 10).

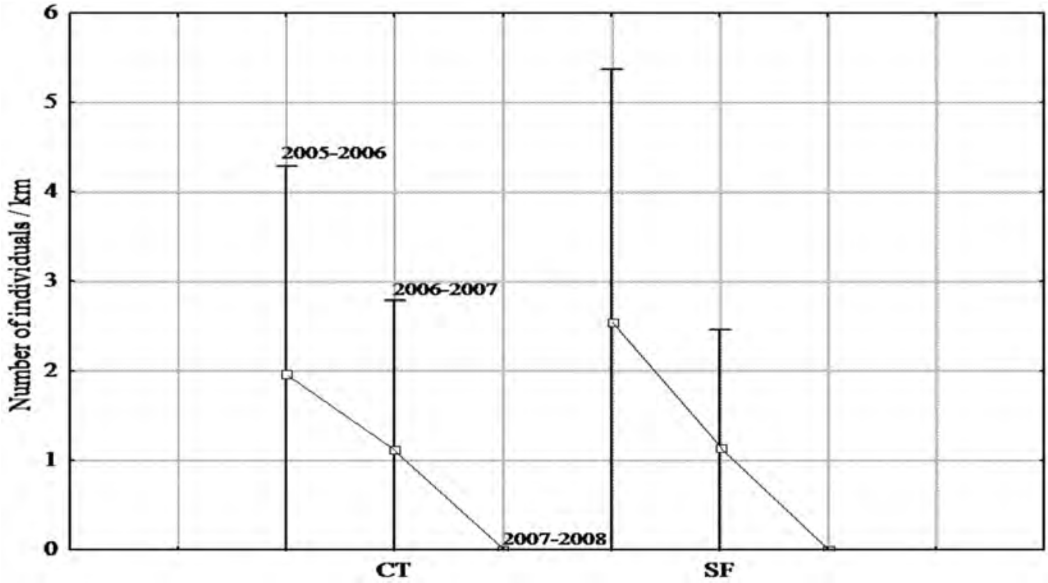


Figure 10. Abundance of Northern Bobwhite (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT], and summer fire treatment [SF] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008.

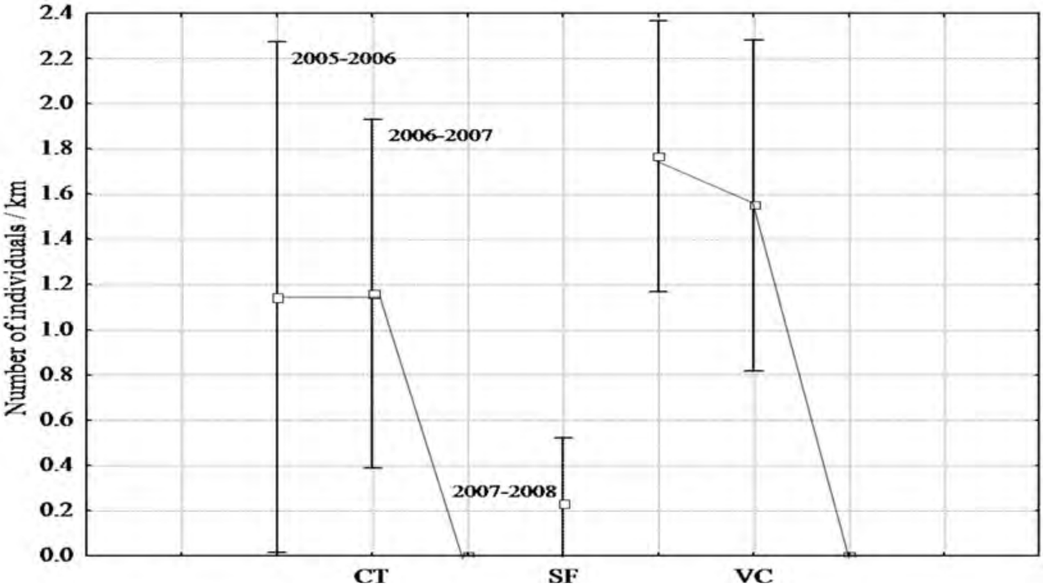


Figure 11. Abundance of Sprague's Pipit (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT] and Vidaurri control [VC] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008, and on summer fire treatment [SF] during winter (Nov 15 to Feb 28) season 2007-2008.

Sprague's Pipit abundance on the combined treatment and Vidaurri control exhibited a similar trend to the other species that declined. Abundance on the combined treatment was similar between the first (1.1 ± 1.2 individuals/km) and second winter season (1.2 ± 0.7 individuals/km), and then went to nearly zero by the third winter (Figure 11). Sprague's Pipit on the Vidaurri control was similar between the first (1.8 ± 0.6 individuals/km) and second winter season (1.6 ± 0.7 individuals/km). But again, by the end of the third winter season, Sprague's Pipit abundance had declined to nearly zero on the combined treatment. Sprague's Pipit was detected on the summer fire treatment only during the third winter season (Figure 11).

Mourning Dove abundance on the combined treatment decreased by 28% from the first winter season (mean ± 95% C.I.; 2.9 ± 3.5 individuals/km) to the second (0.8 ± 0.8 individuals/km), and continued to decrease 90% from its initial abundance by the third winter season (0.3 ± 0.4 individuals/km) . Mourning Dove abundance on the summer fire treatment exhibited no change. Abundance was similar between the first (0.2 ± 0.2 individuals/km) and second winter season (0.03 ± 0.07 individuals/km). By the third winter season, the Mourning Dove abundance had increased by 150% (0.5 ± 0.5 individuals/km). Mourning Dove abundance on the Vidaurri control also was similar during the first (1.4 ± 3.1 individuals/km) and second winter season (9 ± 11.1 individuals/km). However, abundance increased numerically by 643% during this time. By the end of the third winter season, mourning dove abundance had decreased from its initial abundance by 100% (Figure 12).

Sedge Wren abundance on the combined treatment was similar between the first and second winter seasons. Even though 95% confidence intervals overlapped, the mean abundance increased 300% from the first winter season (mean ± 95% C.I.; 0.4 ± 0.4 individuals/km) to the second (1.2 ± 0.8 individuals/km). Sedge Wren abundance had increased significantly from its initial abundance by the third winter season (1.9 ± 0.9 individuals/km), an increase of 475% (Figure 13).

Sedge Wren abundance on the summer fire treatment exhibited a similar trend. Abundance was similar between the first (0.5 ± 0.5 individuals/km) and second winter season (0.6 ± 0.4 individuals/km). By the third winter season, the Sedge Wren abundance had increased from its initial abundance by 280% to 1.4 ± 1.2 individuals/km. Sedge Wren abundance on the Vidaurri control also was

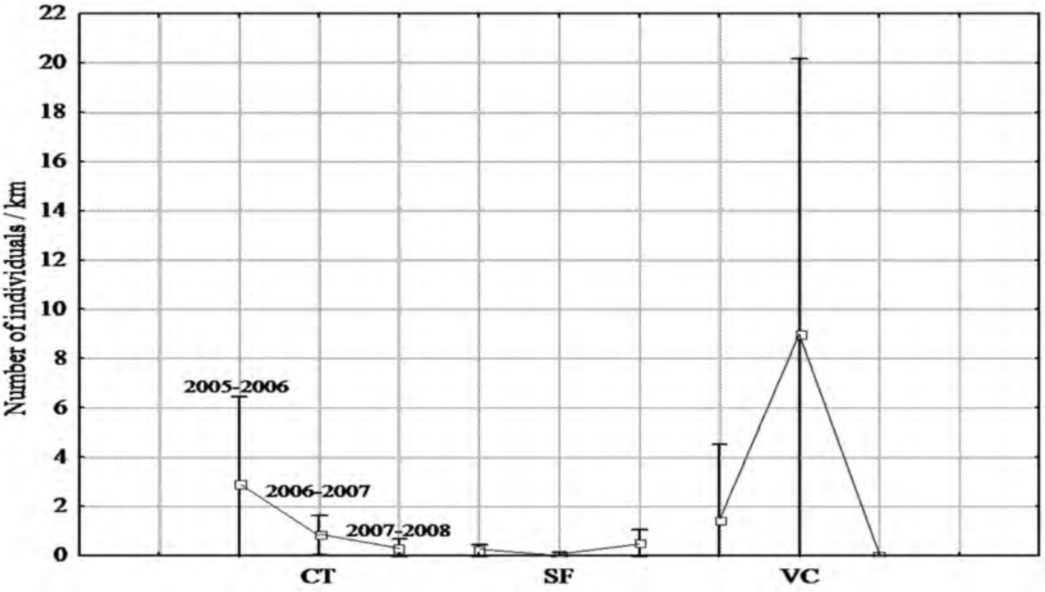


Figure 12. Abundance of Mourning Dove (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT], summer fire treatment [SF], and Vidaurri control [VC] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008.

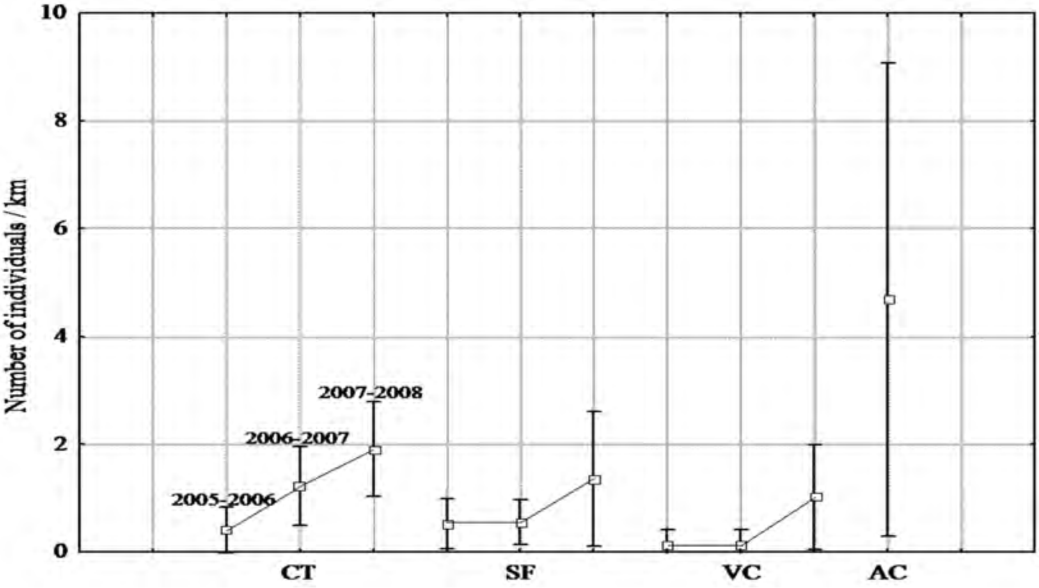


Figure 13. Abundance of Sedge Wren(mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT], summer fire treatment [SF], and Vidaurri control [VC] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008; and on Aransas control [AC] during winter season (Nov 15 to Feb 28) season 2007-2008.

similar during the first (0.1 ± 0.3 individuals/km) and second winter season (0.1 ± 0.2 individuals/km). Sedge Wren abundance increased by the third winter season to 4.7 ± 4.4 individuals/km. Sedge Wren abundance was similar between Vidaurri and Aransas controls (Figure 13).

Sedge Wren abundance exhibited an increasing trend through time across all treatments. The general pattern that emerged was that abundance was similar between the first and the second winter season but had increased by the third winter season. These increases from beginning to ending abundances represented increase of 475%, 280%, and 4,700% on the combined treatment, summer fire treatment, and on the Vidaurri control, respectively (Figure 13).

Eastern Phoebe abundance on the combined treatment was similar between the first, second and third winter seasons. Even though 95% confidence intervals overlapped, the mean abundance increased 71% from the first winter season (mean ± 95% C.I.; 0.7 ± 0.4 individuals/km) to the second (1.2 ± 0.7 individuals/km). Eastern Phoebe abundance had decreased from its initial abundance by the third winter season to 0.3 ± 0.3 individuals/km, a decrease of 57% . On the summer fire treatment, eastern phoebe abundance was similar between the first, second and third winter seasons. Even though 95% confidence intervals overlapped, the mean abundance decreased 50% from the first winter season (mean ± 95% C.I.; 1.8 ± 1.5 individuals/km) to the second (0.9 ± 0.6 individuals/km). Eastern Phoebe abundance had decreased from its initial abundance by the third winter season (0.7 ± 1.3 individuals/km), a decrease of 61% (Figure 14).

Grasshopper Sparrow abundance on the combined treatment increased from the first winter season (0.3 ± 0.4 individuals/km) to the second (4 ± 1.3 individuals/km). By the end of the third winter season, Grasshopper Sparrow abundance had decreased to nearly zero.

The same trend was observed for Grasshopper Sparrows on the summer fire treatment. Grasshopper Sparrow abundance was different between the first and second winter seasons. The mean abundance increased significantly 422% from the first winter season (mean ± 95% C.I.; 0.9 ± 0.6 individuals/km) to the second (3.8 ± 2.1 individuals/km). By the end of the third winter season, Grasshopper Sparrow abundance had decreased significantly from its initial abundance by 100%. Grasshopper Sparrow during the second winter season had an

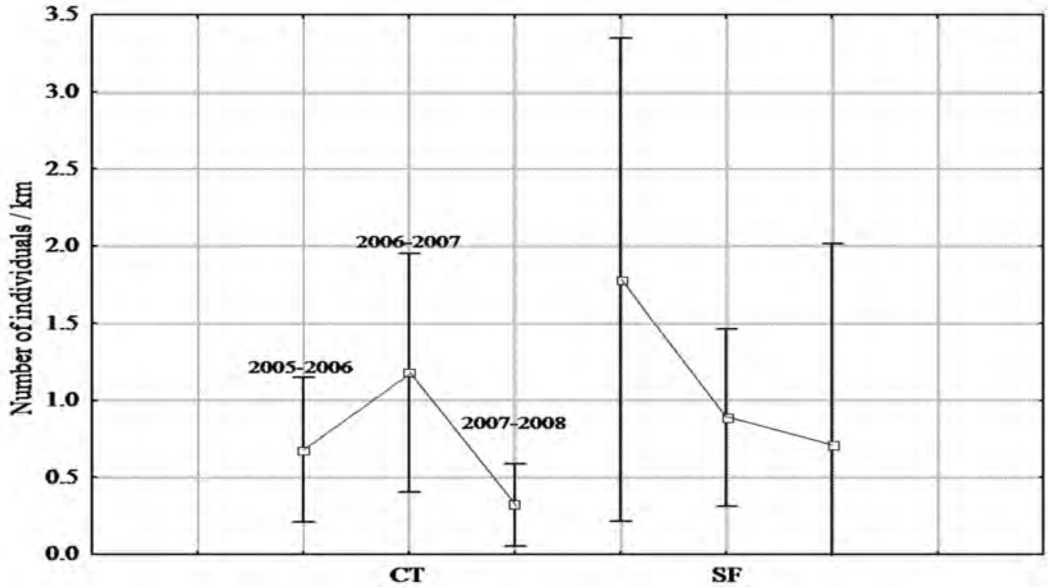


Figure 14. Abundance of Eastern Phoebe (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT] and summer fire treatment [SF] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008.

abundance of (0.98 ± 1.0 individuals/km) and by the third winter season had declined to zero. (Figure 15).

Swamp Sparrow abundance on the combined treatment decreased 31% from the second winter season (mean ± 95% C.I.; 1.6 ± 1.1 individuals/km) to the third (0.5 ± 0.4 individuals/km). Swamp Sparrow abundance on summer fire treatment

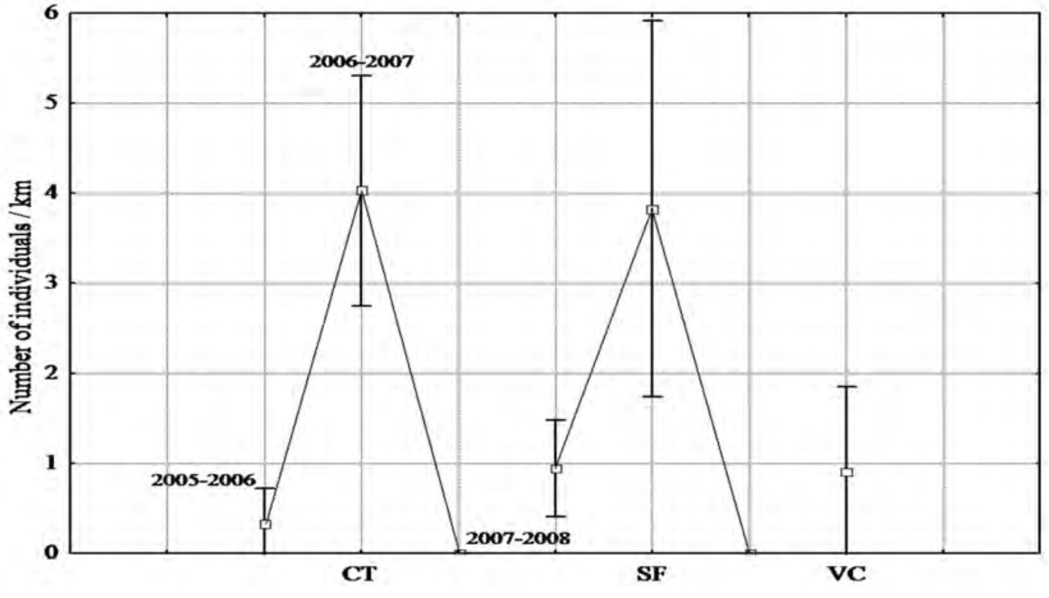


Figure 15. Abundance of Grasshopper Sparrow (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT] and summer fire treatment [SF] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008, and on Vidaurri control [VC] during winter (Nov 15 to Feb 28) season 2006-2007.

had the same trend than on combined treatment, the second and third winter seasons were similar. The mean abundance decreased 80% from the second winter season (mean ± 95% C.I.; 1 ± 0.6 individuals/km) to the third (0.8 ± 1.0 individuals/km). Swamp Sparrow abundance exhibited an up and down trend across the combined and summer fire treatments. The general pattern observed was that during the first winter season was absent, and during the second and third winter season's abundance was statistically similar. Even though 95% confidence intervals overlapped, there was an overall decrease of 69% and 20% on combined and summer fire treatments respectively (Figure 16).

Vegetation

Brush Canopy Cover.—Brush canopy cover on the summer fire treatment was statistically different from the combined treatment. The mean percentage of brush canopy cover on the summer fire treatment (mean ± 95% C.I.; 34.7 ± 3.8) was 3 times greater than on the combined treatment (11.5 ± 2.6).

On the summer fire treatment, mesquite and huizache shrubs represented 100% of the canopy cover composition. Even though the 95% confidence intervals overlapped between mesquite and huizache canopy cover, the mean canopy cover of mesquite (19.3 ± 3.4) was 18% greater than the mean canopy cover of huizache (15.8 ± 2.9). On the combined treatment, mesquite and huizache shrubs represented the 93% of the canopy cover composition. Mesquite and huizache canopy cover was statistically different with mesquite cover (9.8 ± 2.7) being 11 times greater than huizache cover (0.9 ± 0.6). Mesquite and huizache cover was statistically different between summer fire and combined treatments. For example, mesquite and huizache canopy cover was 97% and 17 times greater on the summer fire treatment than on the combined treatment, respectively (Figure 17).

Brush density (plants/25m^2) on the summer fire treatment was statistically different than the combined treatment. The mean brush density on the summer fire treatment (mean ± 95% C.I.; 4.6 ± 0.6) was 2.7 times greater than on the combined treatment (1.7 ± 0.4). On the summer fire treatment, mesquite and huizache shrubs represented 100% of the brush composition. Even though the 95% confidence intervals overlapped between mesquite and huizache brush density, the mean density of mesquite (2.7 ± 0.5) was 30% greater than the mean density of huizache (1.9 ± 0.3) (Figure 18).

On the combined treatment, mesquite and huizache shrubs represented 77% of the brush density composition. Mesquite and huizache density was statistically different. Mesquite density (1.0

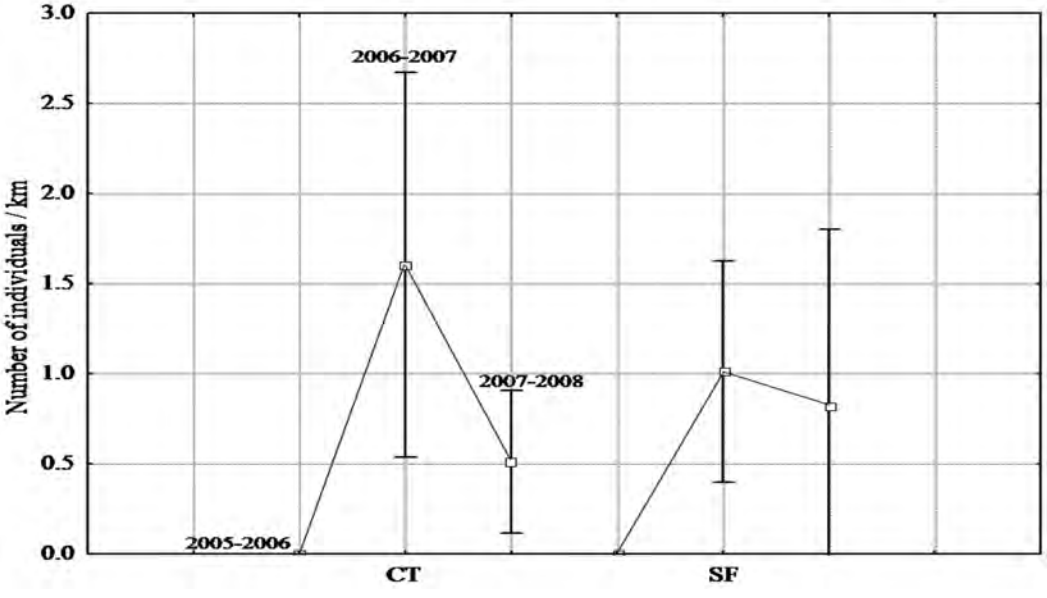


Figure 16. Abundance of Swamp Sparrow (mean number of individuals / km) (Mean ± 0.95 C.I.) on combined treatment [CT] and summer fire treatment [SF] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008.

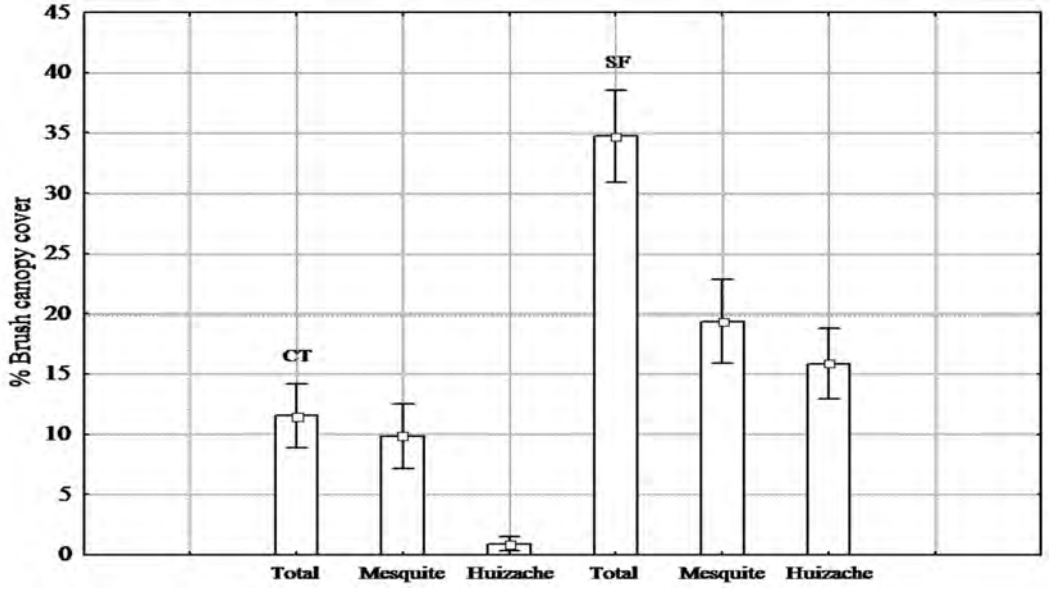


Figure 17. Percentage of total brush canopy cover, mesquite canopy cover, and huisache canopy cover on combined treatment [CT], and summer fire treatment [SF] during the winter season (2005-2006).

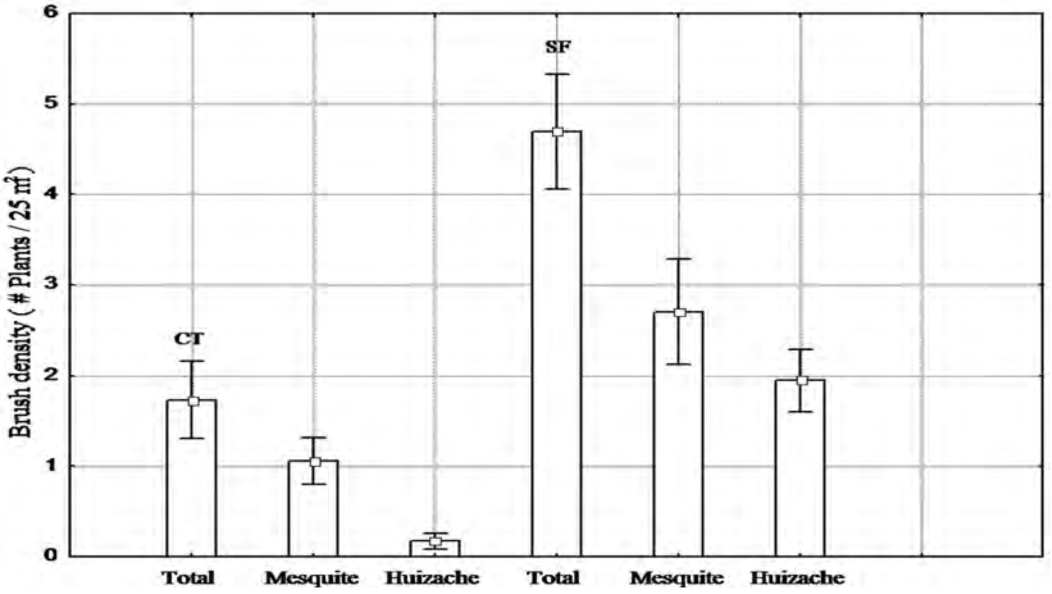


Figure 18. Total brush density, mesquite density, and huizache density on combined treatment [CT], and summer fire treatment [SF] during the winter season (2005-2006).

± 0.3) was 6 times greater than huizache density (0.1 ± 0.09). Density of mesquite and huizache was statistically different between the summer fire treatment and the combined treatment. For example, mesquite and huizache brush density was 3 and 11 times greater on summer fire treatment than on the combined treatment, respectively.

Brush canopy cover on the summer fire treatment, combined treatment, and Aransas control was statistically different. Canopy cover on the summer fire treatment (38.0 ± 5.7) was 4 times greater than the canopy cover on the combined treatment (10.1 ± 2.8), and canopy cover on the combined treatment was 14 times greater than on the Aransas control (0.7 ± 0.4) (Figure 19).

Brush Density.—Brush density on the summer fire treatment, combined treatment, and control sites was statistically different. Brush density on the summer fire treatment (3.5 ± 0.5) was 2, and 35 times greater than on the combined treatment (1.6 ± 0.4) and Aransas control (0.1 ± 0.1), respectively. Aransas control and Vidaurri control brush density were statistically similar. Even though 95% confidence intervals overlapped, the mean brush density on Vidaurri control (0.02 ± 0.06) was 9 times lower compared with mean brush density on Aransas control (Figure 20).

On the combined treatment, brush density by height classes (0-1, 1-2, and >2 meters) was statistically similar. Even though 95% confidence intervals overlapped among density height classes, the mean density of height class >2m (0.6 ± 0.2) was 17%, and 33% greater than the mean density on the height classes 1-2 (0.5 ± 0.2) and 0-1 (0.4 ± 0.1), respectively. On summer fire treatment, the brush density of height class >2 m was statistically greater than on height classes 1-2, and 0-1. The mean brush density of height class >2 m (1.8 ± 0.3) was 2 and 2.5 times greater than the mean density of height classes 1-2 (0.9 ± 0.3), and 0-1m (0.7 ± 0.2), respectively. Brush density by height classes (0-1, 1-2, and >2 meters) on Aransas control was statistically similar. Even though 95% confidence intervals overlapped among density height classes, the mean density of height class 0-1 (0.1 ± 0.07) was 2 and 10 times greater than the mean density of height classes 1-2 (0.05 ± 0.07) and >2 (0.01 ± 0.02), respectively (Figure 21).

Herbaceous Vegetation.—Herbaceous vegetation cover was statistically different during two consecutive winter seasons on the combined treatment plot. During the first winter season, mean grass cover (46.0 ± 2.7) was 2.4 times greater than mean forb cover (19.1 ± 1.5); and the mean litter cover (24.3 ± 2.3) was 23% greater than bare ground cover (18.6 ± 2.1). During the second winter season, the mean of grass cover (72.5 ± 3.2) was 2.4 times greater than the mean forb cover (30.0 ± 2.3); and the mean of litter cover (5.9 ± 1.2) was 3.4 times greater than bare ground cover (1.7 ± 0.6) (Figure 22).

The classes of herbaceous cover were statistically

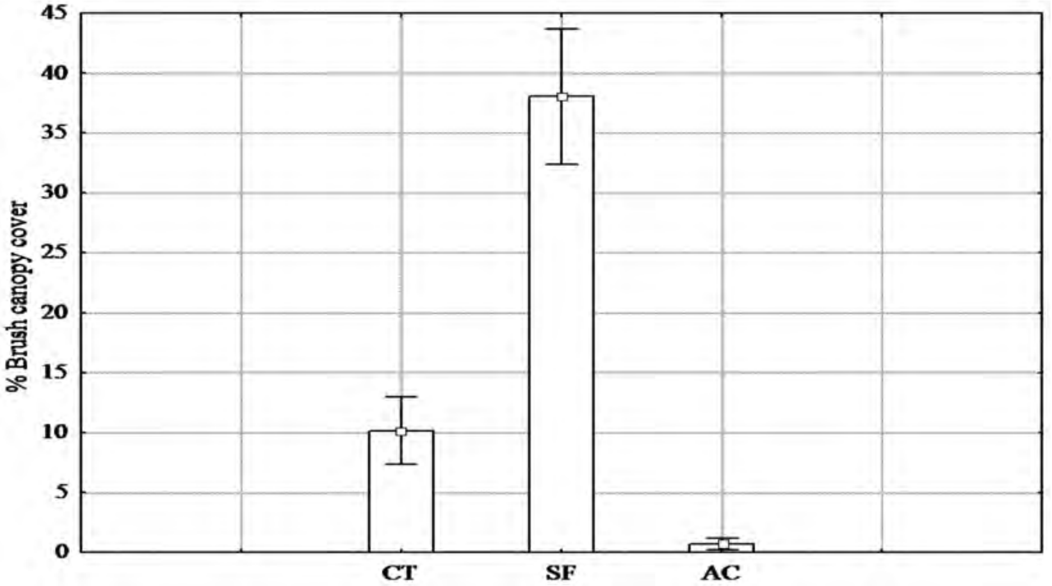


Figure 19. Percentage of brush canopy cover on combined treatment [CT], summer fire treatment [SF], and Aransas control [AC] during the breeding season 2007.

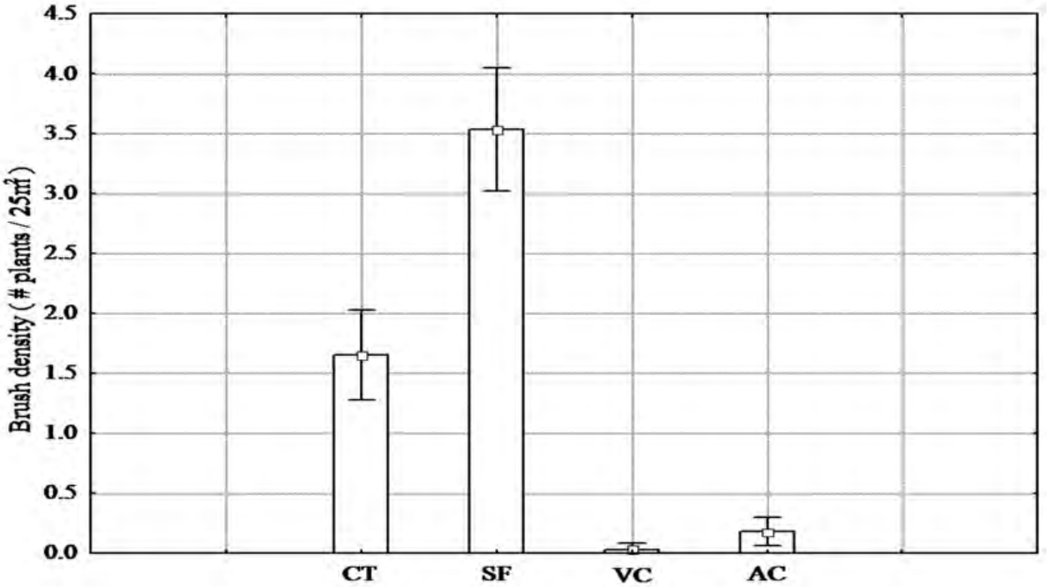


Figure 20. Brush density on combined treatment [CT], and summer fire treatment [SF], Vidaurri control [VC] and Aransas control [AC] during the breeding season 2007.

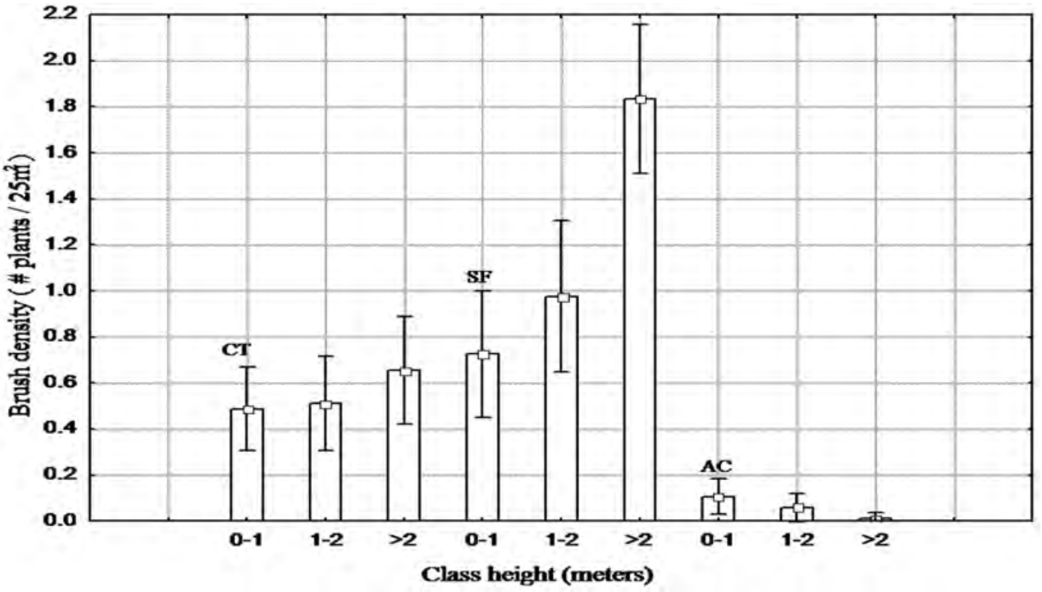


Figure 21. Brush density by height classes (0-1, 1-2, >2 meters) on combined treatment [CT], and summer fire treatment [SF], and Aransas control [AC] during the breeding season 2007.

different during two consecutive winter seasons in the summer fire treatment plot. During the first winter season, the mean of grass cover (58.7 ± 2.9) was 4.6 times greater than the mean of forb cover (12.8 ± 1.0), while the mean of litter cover (26.4 ± 2.2) was 3.2 times greater than bare ground cover (8.1 ± 1.3). The grass and forb cover was statistically different during the second winter season. Grass cover (74.6

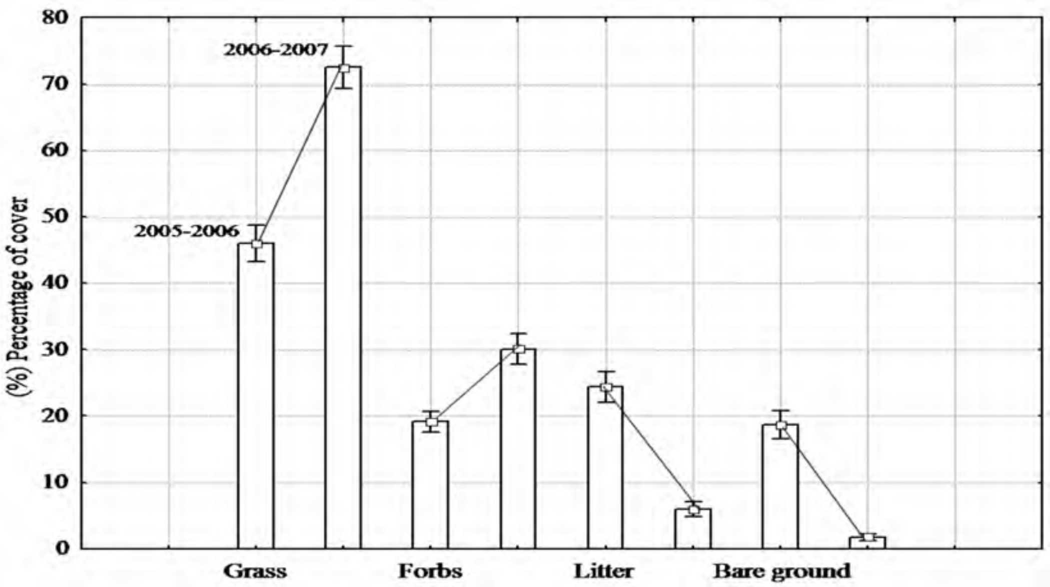


Figure 22. Percentage of grass canopy cover, forb cover, litter cover, and bare ground cover on combined treatment [CT] during two consecutive winter seasons (2005-2006, 2006-2007).

± 2.3) was 2.3 times greater than forb cover (32.5 ± 2.0). On the other hand, the mean of litter (2.3 ± 0.4) and bare ground cover (1.4 ± 0.4), even though the 95% confidence intervals overlapped the mean of litter cover was 39% greater than bare ground. The herbaceous cover was statistically different between seasons. For example, grass and forb cover during the second winter season increased 27% and 2.5 times respectively compared with the first winter season, whereas the mean of litter and bare ground cover during the second winter season decreased 11 and 6 times respectively, compared with the first winter season (Figure 23).

Herbaceous cover was statistically different during the first and second winter seasons on the Vidaurri Control plot. In this regard, grass cover (42.9 ± 3.8) was 4.3 times greater than the mean of forb cover (9.8 ± 1.1), while litter cover (39.7 ± 3.1) was 2.9 times greater than bare ground cover (13.5 ± 1.9).

During the second winter season, grass and forb cover was statistically different. Grass cover (73.3 ± 3.5) was 3.1 times greater than forb cover (23.6 ± 2.4), whereas litter cover (6.8 ± 1.0) and bare ground cover (4.8 ± 2.2) was statistically similar during the second winter season. Even though 95% confidence intervals overlapped between litter and bare ground cover, litter cover was 29% greater than bare ground cover (Figure 24).

Herbaceous cover within classes on the Vidaurri control was statistically different between seasons. Grass and forb cover during the second winter season increased 70% and 2.4 times, respectively, compared with the first winter season; while litter and bare ground cover during the second winter season decreased 5.8 and 2.8 times respectively compared with the first winter season (Figure 24).

Grass Height.—Grass height on the combined treatment, summer fire treatment and Vidaurri control was statistically different during two consecutive winter seasons. During the first winter season, grass height (23.3 ± 1.4 cm) on summer fire treatment was 22% greater than on the combined treatment (19.1 ± 1.3), and the combined treatment was 18% greater than on the Vidaurri control (15.6 ± 1.5). During the second winter season, grass height on the combined treatment (45.9 ± 2.7) was 12% greater than on the summer fire treatment (40.3 ± 1.8), whereas the summer fire treatment was 12% greater than on the Vidaurri control (35.5 ± 2.4; Figure 25).

Grass height within combined treatment, summer fire treatment, and Vidaurri control was statistically different between seasons. On combined treatment

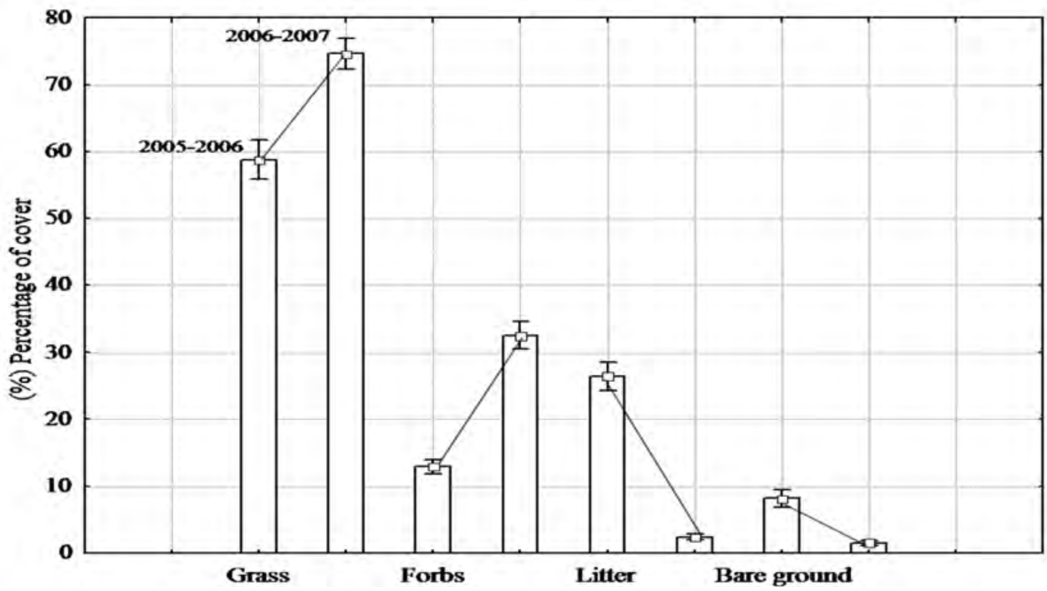


Figure 23. Percentage of grass canopy cover, forb cover, litter cover, and bare ground cover on summer fire treatment [SF] during two consecutive winter seasons (2005-2006, 2006-2007).

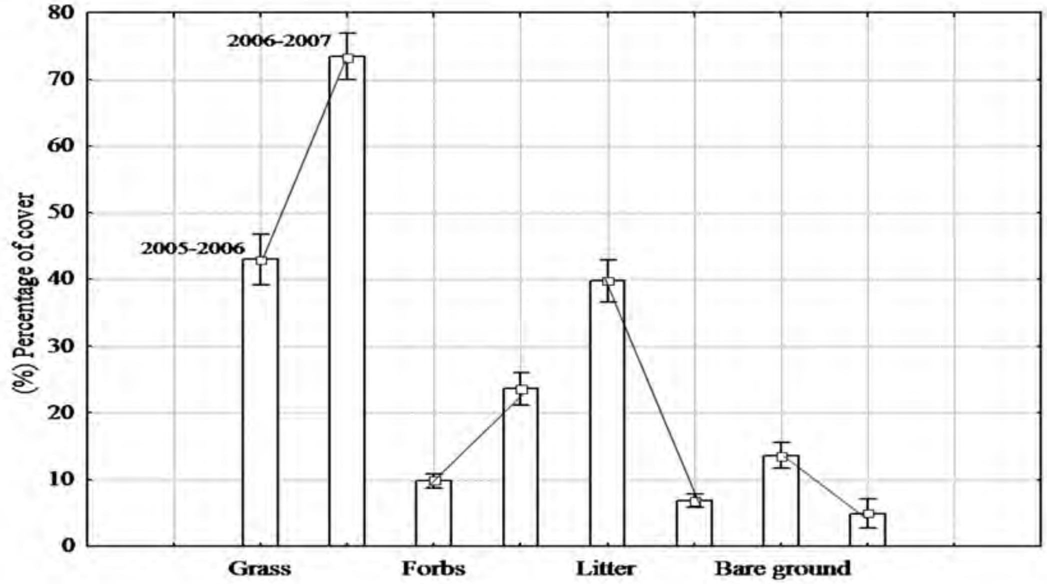


Figure 24. Percentage of grass canopy cover, forb cover, litter cover, and bare ground cover on Vidaurri control [VC] during two consecutive winter seasons (2005-2006, 2006-2007).

and Vidaurri control, grass height increased 2.4 and 2.3 times during the second winter season compared with the first winter season, respectively; and during the second winter season on the summer fire treatment, increased 73% compared with the first winter season (Figure 25).

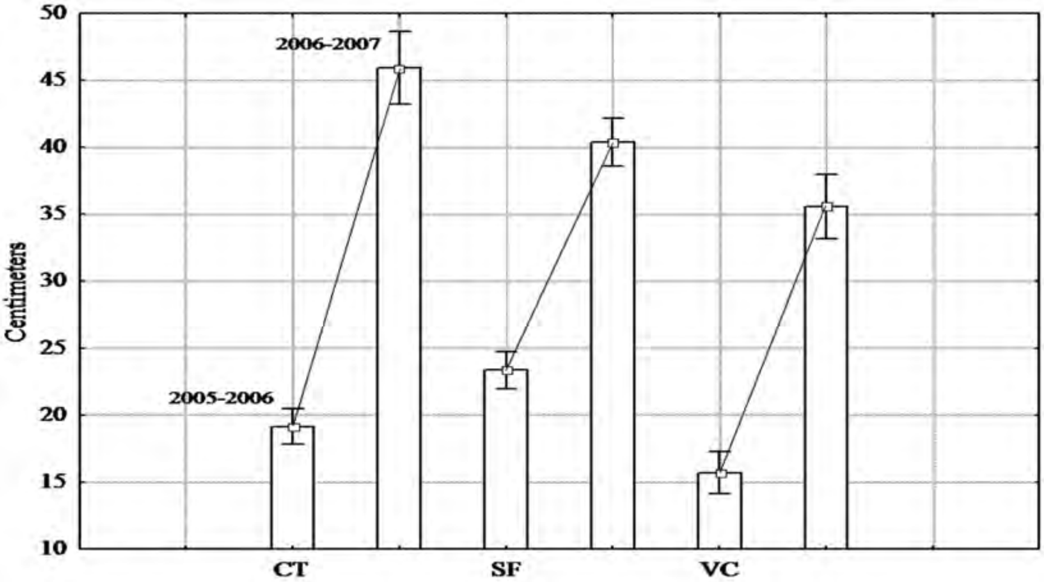


Figure 25. Grass height (cm) on combined treatment [CT], summer fire treatment [SF], and Vidaurri control [VC] during two consecutive winter seasons (2005-2006, 2006-2007).

DISCUSSION

Breeding and Wintering Bird Species Richness

Breeding bird species richness was consistently greater on the combined and summer fire treatments compared with the Vidaurri control during two consecutive breeding seasons, and during the second breeding season at the Aransas control. Both treatments had at least 40%, and 60% greater species richness than the Vidaurri control respectively; and 20% greater species richness than on Aransas control. Wintering bird species richness also followed a similar pattern, with richness being consistently higher on the combined treatment and sum fire treatment plots compared to the controls, which was consistent with our research hypothesis.

Vegetation structure was apparently a major influence on the differences in species richness between the control plots and the treatment plots. Increasing diversity in vegetation structure has long been known to have a strong, positive relationship with bird species diversity (MacArthur 1964, MacArthur et al. 1966, Tramer 1969, Karr and Roth 1971, Tomoff 1974). Tramer (1969) pointed out that grasslands only have one vertical layer of vegetation compared with at least 3 vertical layers in shrublands. Tomoff (1974) documented higher species richness in relation to an index of foliage height diversity. In this regard, our results were similar to those of Tramer (1969) and Tomoff (1974). On our areas brush abundance by height classes (0-1, 1-2, >2 m) was much greater on both the combined and summer fire treatments plots compared with the Vidaurri control which did not have any shrubs. The Aransas control contained very few shrubs that were scattered throughout the landscape. Willson (1974) found that two indexes of vegetation structure (foliage height diversity and percent of vegetation cover) were related to bird species diversity, which also corresponded to our results. The brush component throughout the landscape on treatments and control sites played an important role by providing perch sites for calling, courtship, and nest building for some species, and ultimately structural diversification of habitat. In a study during the winter in south Texas, Emlen (1972) found that grasslands had lower avian species richness compared to shrubland habitats. He reported 14 bird species in grasslands and 35 species on shrublands. His grassland site was comparable to the combined treatment plots in this study.

Species richness within treatments decreased as time progressed. The more abrupt change from the first winter season to the third and last winter season was on the combined and summer fire treatments.

In the case of the combined treatment it had initially 15 species and then decreased to 12 species by the end of the study, while summer fire treatment had 31 species during the first winter season but decreased to 13 species on the last winter season. The decline of species richness was less apparent on the Vidaurri control with 7 species which were detected during the first winter season and then decreased to 5 on the last winter season.

Brush cover and density on both combined and summer fire treatments played an important role by providing a more diverse vegetation structure than on control sites during the first two winter seasons. Hence, it provided more structural diversification for birds habitat, it seemed there was a minimum species richness attained in these treatments which was between 12 and 13 species. Similar findings were observed by Roth (1976) who found between 13 and 15 bird species in the coastal bend in south Texas, and concluded that about 15 species is the community norm for this region. More recently Igl and Ballard (1999) found that species richness was lower on grasslands than on shrublands. They found 25 bird species in grasslands compared with 46 in shrublands, however, the grassland mentioned in their study was composed of 10% of brush canopy cover which coincides with the vegetation structure of the combined treatment in this study (12%). The 25 species found by Igl and Ballard on their "grassland habitat" was similar to the species richness found during the second winter season in both combined and summer fire treatments, which had 25 and 24 species, respectively.

Breeding Bird Abundance

Along with Cliff Swallow, Eastern Meadowlark and Northern Mocking bird were the most abundant breeding bird species across treatments and seasons. In the case of the eastern meadowlark---a grassland obligate species that is declining range wide---abundance was greater on control sites than on the combined treatment site. Similar results were found by Grant et al. (2004), who observed that meadowlarks decreased as tall shrub cover increased, a situation which was only found in the summer fire treatment. Warren and Anderson (2005) found that Eastern Meadowlarks preferred sites for nesting with 12% of standing dead vegetation and a vegetation height of about 56 cm with no brush component. In another study, Haroldson et al. (2006) found that Eastern Meadowlarks used areas with short dense grass on sites that were lightly grazed.

Wintering Bird Abundance

On the combined treatment plot bare ground cover was about 21% during the first winter season and then declined abruptly to about 2% during the second winter season, a similar pattern occurred on the summer fire treatment plot with about 10% during the first winter season and then declined to less than 2% on the second season. By the third winter season bare ground was absent from both treatments, and this coincided with an absence of Northern Bobwhites on the same year. The absence of the Northern Bobwhite on control sites can be attributed to the lack of brush canopy cover (Guthery 1986, Fulbright and Guthery 1995). Grzybowski (1982) found that savanna sparrow was the dominant species on lightly grazed sites. These findings were similar to our study, at least during the first two winter seasons where abundance was significantly greater than the third season on both combined and summer fire treatments.

Baldwin et al. (2007) found that Savanna Sparrow abundance was highest on areas 1 year post burn with sparse herbaceous vegetation and diverse shrub density. These previous findings correspond with our results during the first winter season on the combined and summer fire treatments. Those treatments had bare ground and sparse herbaceous vegetation and had a greater abundance of Savanna Sparrow during the first winter season on the combined and summer fire treatment. Grzybowski (1982) found that Eastern Meadowlark was more abundant on heavy grazed sites compared with the moderately grazed sites, which was similar to our study. Coppedge et al. (2008) found that Eastern Meadowlark was related positively with tallgrass, shortgrass and litter cover. Lueders et al. (2006) found that Savanna Sparrow abundance was 4.9 times greater on plots grazed by cattle than by bison and recommend the use of moderate stocking rate to provide lower vegetation structure. Grzybowski (1983) found that Savanna Sparrow avoided areas with vegetation heights greater than 1 meter, similar to our findings in both treatments where the grass height was below 1 meter during the first two winter seasons. Mourning Dove is another species that needs a bare ground component in its habitat.

For example, Ostrand et al. (1996) found that Mourning Doves preferred manipulated habitats on conservation reserve program (CRP) fields where the grass height and cover were modified. Baker and Guthery (1990) found that Mourning Doves had higher densities on sites that were heavy grazed. Similar results were found by Lauders et al. (2006). They found that Mourning Dove abundance was higher on burned plots compared with unburned plots. Coppedge et al. (2008) also found that Mourning Dove abundance was positively related with bare ground. Previous findings on Mourning Dove were similar than in our study because the decreasing trend of this species was present along with the decrease and eventual elimination of bare ground as time progressed on the treatments and control site. Davis et al. (1999) found that Sprague's Pipit occurred less frequently in heavily grazed pastures, and was typically found in medium grazed grasslands (NRCS 1999) . In this regard, the grass height on both treatments and control sites increased significantly from the beginning to the end of the study, which could promote Sprague's Pipit.

Only Sedge Wren abundance increased from the beginning of the study to the end across treatments and seasons. Based on its habitat requirements, tall grass vegetation is required for this species (NRCS 1999). This corresponds with the increase of grass cover and height that was present on both treatments and control sites across seasons. The previous pattern was triggered by the great amount of rainfall during the previous summer seasons. Baldwin et al. (2007) found that Sedge Wren was common on 2 year post burn areas composed by dense herbaceous vegetation. This corresponded with our findings because the grass cover and height increased significantly throughout the study.

Grasshopper Sparrow and Swamp Sparrow were the only two species that fluctuated across treatments and seasons. Even though Grasshopper Sparrow has been positively related with dense grass cover and negatively with bare ground cover (Rotenberry and Wiens 1980) its abundance did not increase as expected based on the cumulative increase over time in terms of grass cover and height along with the increment of vertical obstruction in the class height 0.0-0.5 m. Pulliam and Mills (1977) found that Grasshopper Sparrow preferred open grasslands with scattered brush and were usually found between 8 and 16 meters from the shrub cover. Wiens (1973) found that Grasshopper Sparrows were more abundant on ungrazed plots which had zero percent bare ground, 95% of grass cover and 20% of forb cover compared with grazed sites. Grzybowski (1983) mentioned that Grasshopper Sparrow was found on moderately grazed grasslands. Similar results were found by Bock et al. (1984) who found the Grasshopper sparrow on an ungrazed treatment. Regarding the Swamp Sparrow, Baldwin et al. (2007) reported that it was found in areas dominated by dense brush vegetation, and no relationship with herbaceous structure was found. Similar results were found in this study where Swamp Sparrow was not present during the first winter season at the combined and summer fire treatments, however these treatments had a brush component compared with the control site. Tom Langshield (pers. comm.) suggested that the sudden presence of the Swamp Sparrow during the second winter season could be related with the greater amount of rainfall received during the preceding summers, which was above average. Moreover, Baldwin et al. (2007) did not find any relationship between the presence of the Swamp Sparrow and the time since prescribed fires were applied. This provides strong evidence that this species is not related with the lower structure of the vegetation, and possibly may be more closely related to weather events during the previous winter.

Eastern Phoebe also fluctuated in abundance across seasons. Since eastern phoebe is an aerial forager with insects as diet (Canterbury et al. 2000), we did not expect changes on its abundance on the combined and summer fire treatments which provided foraging substrate (brush canopies) for feeding.

Changes in Vegetation

At the beginning of the study there was a clear differentiation in terms of vegetation structure among the treatments and the control sites. The most obvious feature in each treatment was the amount of brush interspersion. The combined treatment was a grassland-savanna with brush patches scattered throughout, and in the middle of them there were small areas of open grasslands with a very few shrubs. For the summer fire treatment, the shrub component was more evenly distributed with less open areas dominated by herbaceous vegetation.

The patchy brush pattern in the combined treatment was less apparent in the summer fire treatment, where shrubs of different heights dominated the landscape. On the other hand, the control sites, which were dominated by grass and co-dominated by forbs had nearly no brush component in them. In descending order, structurally more complex treatments were: the combined treatment, summer fire treatment, and Vidaurri control during the first year of the study, and bare ground as a prominent feature in terms of horizontal heterogeneity at all experimental sites.

During the second year of the study, the herbaceous vegetation grew rapidly and became much denser than the previous year. The most obvious characteristic at all treatment and control sites was the abrupt decline of bare ground, which was almost absent by the third year, giving the impression of an even layer of grass beneath the shrub community. By the third year of the study, the horizontal heterogeneity provided by some gaps in the herbaceous vegetation strata had disappeared. In summary, there was a progressive loss of habitat heterogeneity as time progressed after the brush management treatments were applied. This loss of heterogeneity was directly related to a rapid growth of grasses and forbs that eliminated a matrix of bare ground interspersed among both woody and herbaceous vegetation.

It is important to note that our study area was located in the wettest portion of the south Texas ecoregion with about 34 inches of precipitation annually compared with the 17 inches of rainfall that characterize the western side of south Texas ecoregion (Fulbright and Bryant 2002, Norwine and John 2007), thus, this region has been recognized to have high productivity in terms of forage production (Spears et al. 1993). In this regard, the relatively rapid, positive responses of herbaceous vegetation were expected over the time since treatments were applied.

Roller-chopping, fire and herbicides have different levels and types of effectiveness for reducing brush. Fire will modify habitat structure depending on its frequency (Mushinsky and Gibson 1991) and season (Scifres and Hamilton 1993) and it can increase the heterogeneity of the vegetation in terms of distribution, age, and height. The principal function of roller-chopping as a mechanical means is the top removal of the brush; it also provides seedbed preparation, which promotes water infiltration and seed germination (Hamilton et al. 2004). After an herbicide application, there is an increase of forb production 2 to 5 years post treatment depending on the regional weather and climatic conditions, but forb production is usually reduced during the first year, and recovers by the second growing season. (Hamilton et al. 2004) stated that the synergism effect of a combination of mechanical treatments, fire and herbicides is more effective at controlling shrubs.

Rogers et al. (2004) evaluated the response of sites treated first with an aerator and then maintained by another pass of the same treatment, followed by fire, during the second year of the initial treatment. Brush cover decreased from 22% to 17% on sites treated twice with aerator and from 24% to 14% on sites treated with aerator and fire for maintenance, with a 10% difference between means. Even though these results are from a xeric area in the southwestern Texas plains, they are similar to our results. There seems to be a synergy between roller-chopping, fire and herbicides treatment keeps brush canopy cover around a value of about 12% canopy cover compared with 34% brush cover on the summer fire treatment.

ACKNOWLEDGMENTS

Tom Langshied provided training on bird identification, and Tim Anderson helped with organizing the field plots and other aspects of this project. Eva de Leon provided administrative assistance. Tim Fulbright and Alfonso “Poncho” Ortega provided support and advice at the Caesar Kleberg Wildlife Research Institute. Funding for this project was provided by the Natural Resources Conservation Service and by a scholarship from CONACYT. Patricio Cueva Zambrano also provided support and encouragement. We are indebted to the owners of Vidaurri ranch for access to their property. Leonard Brennan was supported by the C.C. Charlie Winn Endowed Chair and Fidel Hernández was supported by the Alfred Glassell, Jr. Endowed Professorship in the Richard M. Kleberg, Jr. Center for Quail Research. This is publication number 17-110 from the Caesar Kleberg Wildlife Research Institute.

LITERATURE CITED

Askins, R. A. 2000. Restoring North America's birds: lessons from landscape ecology. Yale University Press, New Haven.

Baker, D. L., and F. S. Guthery. 1990. Effects of continuous grazing on habitat and density of ground-foraging birds in south Texas. Journal of Range Management 43:2-5.

Baldwin, H. Q., J. B. Grace, W. C. Barrow, Jr., and F. C. Rohwer. 2007. Habitat relationships of birds overwintering in a managed coastal prairie. The Wilson Journal of Ornithology 119:189-197.

Bock, C. E., J. H. Bock, W. R. Kenney, and V. M. Hawthorne. 1984. Responses of birds, rodents, and vegetation to livestock exclosure in a semidesert grassland site. Journal of Range Management 37:239-242.

Brennan, L. A., and W. P. Kuvlesky, Jr. 2005. North American grassland birds: an unfolding conservation crisis. Journal of Wildlife Management 69:1-13.

Canfield, R. H. 1941. Application of line interception in sampling range vegetation. Journal of Forestry 39: 388-394.

Canterbury, G. E., T. E. Martin, D. R. Petit, L. J. Petit, and D. F. Bradford. 2000. Bird communities and habitat as ecological indicators of forest condition in regional monitoring. Conservation Biology 14: 544-558.

Coppedge, B. R., S. D. Fuhlendorf, W. C. Harrell, and D. M. Engle. 2008. Avian community response to vegetation and structural features in grasslands managed with fire and grazing. Biological Conservation 141:1196-1203.

Davis, S. K., Duncan, D. C., and M. Skeel. 1999. Distribution and habitat associations of the three endemic grassland songbirds in southern Saskatchewan. The Wilson Bulletin 11 l: 389-396.

Daubenmire, R. F. 1959. Canopy coverage method of vegetation analysis. Northwest Science 33:43–64.

Emlen, J. T. 1972. Size and structure of a wintering avian community in southern Texas. Ecology 53:317-329.

Fulbright, T. E., and F. C. Bryant. 2002. The last great habitat. Special Publication Number \1. Caesar Kleberg Wildlife Research Institute, Kingsville, Texas.

Fulbright, T. E., and F. S. Guthery. 1995. Mechanical manipulation of plants. Pages 339-354 in Rangeland wildlife (P.R. Krausman, Editor). Rangeland Wildlife. The Society of Range Management, Denver, Colorado.

Guthery, F. S. 1986. Beef, brush and bobwhites: quail management in cattle country. Caesar Kleberg Wildlife Research Institute Press, Texas A&M University–Kingsville.

Grzybowski, J. A. 1982. Population structure in grassland bird communities during winter. The Condor 84:137-152

Grant, T. A., E. Madden, and G. B. Berkey. 2004. Tree and shrub invasion in northern mixed-grass prairie: implications for breeding grassland birds. Wildlife Society Bulletin 32:807-818.

Gutzwiller, K. J. 1991. Estimating winter species richness with unlimited-distance point counts. The Auk 108:853-862.

Hamel, P. B., W. P. Smith, D. J. Twedt, J. R. Woehr, E. Morris, R. B. Hamilton, and R. J. Cooper. 1996. A land manager's guide to point counts of birds in the Southeast. U.S. Department of Agriculture, Forest Service General Technical Report 50-120.

Hamilton, W. T., A. McGinty, D. N. Ueckert, C. W. Henselka, and M. R. Lee. 2004. Brush management. Past, present, future. Texas A&M University Press, College Station.

Haroldson, K. J., R. O. Kimmel, M. R. Riggs, and A. H. Berner. 2006. Association of ring-necked pheasant, gray partridge, and meadowlark abundance to conservation reserve program grasslands. Journal of Wildlife Management 70:1276-1284.

Hutto, R. L., S. M. Pletschet, and P. Hendricks. 1986. A fixed-radius point count method for nonbreeding and breeding season. The Auk 103:593-602.

Igl, L. D. and B. M. Ballard. 1999. Habitat association of migrating and overwinter grassland birds in South Texas. The Condor 101:771-782.

Johnson, D. H. 1999. The insignificance of statistical significance testing. Journal of Wildlife Management 63:763-772.

Karr, J. R., and R. R. Roth. 1971. Structure of avian communities in selected Panama and Illinois habitats. Ecological Monographs 41:207-233.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 5:247-257.

Lauders, A. S., P. L. Kennedy, and D. H. Johnson. 2006. Influences of management regimes on breeding bird densities and habitat in mixed-grass prairie: an example from North Dakota. The Journal of Wildlife Management 70:600-606.

Lozano-Cavasos, A. E. 2009. Avian community response to brush reduction in Texas Coastal Prairie habitats. Dissertation. Texas A&M University–Kingsville.

MacArthur, R. H. 1964. Environmental factors affecting species diversity. American Naturalist 98:387-398.

MacArthur, R. H., H. R. Recher, and M. Cody. 1966. On the relation between habitat selection and species diversity. American Naturalist 100:319-322.

Morrison, M. L., W. M. Block, M. D. Strickland and W. L. Kendall. 2001. Wildlife study design. Springer-Verlag, New York.

Mushinsky, H. R. AND D. J. Gibson. 1991. The influence of fire periodicity on habitat structure. Pp. 237-259 in Habitat structure (Bell, S. S. et al. Editors). Chapman and Hall. London, United Kingdom.

Norwine, J and K. John. 2007. The Changing Climate of South Texas 1900-2100. Texas A&M University–Kingsville, Kingsville.

NRCS. 1999. Grassland birds. Fish and wildlife habitat management leaflet. Number 8.

Ostrand, W. D., Bissonette, J. A., and M. R. Conover. 1996. Selection of foraging habitat by mourning doves: a structural approach. Journal of Field Ornithology 67:691-697.

Peterson, M. J., X. B. Wu, and P. Rho. 2002. Rangewide trends in land use and Northern Bobwhite abundance: an exploratory analysis. Proceedings of the National Quail Symposium. 5:35-44.

Pulliam, H. R., and G. S. Mills. 1977. The use of space by wintering sparrows. Ecology 58:1393-1399.

Rogers, J. O., T. E. Fulbright, and D. C. Ruthven III. 2004. Vegetation and deer response to mechanical shrub clearing and burning. Journal of Range Management 57:41-48.

Rotenberry, J. T., and J. A. Wiens. 1980. Habitat structure, patchiness, and avian communities in North American steppe vegetation: a multivariate analysis. Ecology 61:1228-1250.

Roth, R. R. 1976. Spatial heterogeneity and bird species diversity. Ecology. 57:773-782.

Samson, F. and F. Knopf. 1994. Prairie conservation in North America. Bioscience 44:418-421.

Scifres, C. J., and W. T. Hamilton. 1993. Prescribed burning for brushland management: the south Texas example. Texas A&M University Press, College Station.

Spears, G. S., F. S. Guthery, S. M. Rice, S. J. DeMaso, and B. Zaiglin. 1993. Optimal seral stage for northern bobwhites as influenced by site productivity. Journal of Wildlife Management 57:805-811.

Tomoff, C. S. 1974. Avian species diversity in desert scrub. Ecology 55:396-403.

Tramer, E. J. 1969. Bird species diversity: components of Shannon's formula. Ecology 50:927-929.

USDA-NRCS. 2008. Web Soil Survey. http://websoilsurvey.nrcs.usda.gov/app/ (accessed August 2, 2008).

Warren, K. A., and J. T. Anderson. 2005. Grassland songbird nest-site selection and response to mowing in West Virginia. Wildlife Society Bulletin 33:285-292.

Wiens, J. A. 1973. Pattern and process in grassland bird communities. Ecological monographs 43: 237-270.

Willson, M. F. 1974. Avian community organization and habitat structure. Ecology 55:1017-1029.

Appendix 1. Common and scientific names of birds, mammals and plants mentioned in the text.

Common name	Scientific name
Birds	
American Kestrel	*Falco sparverius*
American Pipit	*Anthus rubescens*
American Robin	*Turdus migratorius*
Ash-throated Flycatcher	*Myiarchus cinerascens*
Bewick's Wren	*Thryomanes bewickii*
Black-throated Sparrow	*Spizella atrogularis*
Blue-gray Gnatcatcher	*Polioptila caerulea*
Blue Grosbeak	*Passerina caerulea*
Bobolink	*Dilochonyx oryzivorus*
Burrowing Owl	*Athene cunicularia*
Brewer's Sparrow	*Spizella breweri*
Brown-crested Flycatcher	*Myiarchus tyrannulus*
Brown-headed Cowbird	*Molothrus ater*
Crested Caracara	*Caracara cheriway*
Carolina Chickadee	*Poecile carolinensis*
Cassin's Sparrow	*Aimophila cassinii*
Cattle Egret	*Bubulcus ibis*
Cave Swallow	*Petrochelidon fulva*
Chipping Sparrow	*Spizella passerina*
Clay-colored Robin	*Turdus grayi*
Clay-colored Sparrow	*Spizella pallida*
Cliff Swallow	*Petrochelidon pyrrhonota*
Common Nighthawk	*Chordeiles minor*
Dickcissel	*Spiza americana*
Eastern Bluebird	*Sialia sialis*
Eastern Meadowlark	*Sturnella magna*
Eastern Phoebe	*Sayornis phoebe*
Field Sparrow	*Spizella pusilla*
Golden-fronted Woodpecker	*Melanerpes aurifrons*
Grasshopper Sparrow	*Ammodramus savannarum*
Horned Lark	*Eremophila alpestris*
House Wren	*Troglodytes aedon*
Henslow's Sparrow	*Ammodramus henslowii*
Inca Dove	*Columbia inca*
Ladder-backed Woodpecker	*Picoides scalaris*
Lark Bunting	*Calamospiza melanocorys*

Appendix 1. (*continued*).

Common name	Scientific name
Birds	
Lark Sparrow	*Chondestes grammacus*
LeConte's Sparrow	*Ammodramus leconteii*
Lincoln's Sparrow	*Molospiza lincolnii*
Loggerhead Shrike	*Lanius ludovicianus*
Lucy's Warbler	*Vermivora virginiane*
Mourning Dove	*Zenaida macroura*
Northern Bobwhite	*Colinus virginianus*
Northern Cardinal	*Cardinalis cardinalis*
Northern Mockingbird	*Mimus polyglottos*
Painted Bunting	*Passerina ciris*
Pyrrhuloxia	*Cardinalis sinuatus*
Red-shouldered Hawk	*Buteo lineatus*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Slackbird	*Agelaius phoeniceus*
Ruby-crowned Kinglet	*Regulus calendula*
Sage Sparrow	*Amphispiza belli*
Savanna Sparrow	*Passerculus sandwichensis*
Scissor-tailed Flycatcher	*Tyrannus forficatus*
Sedge Wren	*Cistothorus platensis*
Song Sparrow	*Melospiza melodia*
Sprague's Pipit	*Anthus spragueii*
Swamp Sparrow	*Molospiza georgiana*
Vermilion Flycatcher	*Pyrocephalus rubinus*
Vesper Sparrow	*Poecetes gramineus*
Western Meadowlark	*Sturnella neglecta*
White-crowned Sparrow	*Zonotrichia leucophrys*
White-tailed Hawk	*Buteo albicaudatus*
White-tailed Kite	*Elanus leucurus*
Yellow-billed Cuckoo	*Coccyzus americanus*
Mammals	
Bison	*Bos bison*
White-tailed deer	*Odocoileus virginianus*
Plants	
Huizache (Huisache)	*Acacia farnesiana*
Mesquite	*Prosopis glandulosa*

Appendix 2. Number of detections / point count / of all bird species detected on on combined treatment [CT], summer fire treatment [SF], Vidaurri control [VC], and on Aransas control [CA] during two consecutive breeding (May 15 to July 31) seasons 2006 and 2007.

Species	Number of detections / point count						
	First season			Second season			
	CT	SF	VC	CT	SF	VC	AC
Ash-throated Flycatcher	—	—	—	—	0.04	—	—
Blue Grosbeak	0.04	0.2	—	—	—	—	—
Brown-crested Flycatcher	—	0.1	—	—	0.5	—	—
Brown-headed Cowbird	0.1	0.2	—	—	0.04	—	—
Cassin’s Sparrow	0.04	0.2	0.04	—	—	—	—
Carolina Chickadee	—	0.2	—	0.01	—	—	—
Cattle Egret	—	—	0.04	—	—	—	—
Cave Swallow	—	0.1	—	—	—	—	—
Clay-colored Robin	—	—	—	0.01	—	—	—
Cliff Swallow	0.9	0.1	1.1	3.1	—	—	0.8
Common Nighthawk	0.04	—	—	—	—	—	—
Dickcissel	—	—	—	0.6	0.5	0.9	0.5
Eastern Meadowlark	0.1	—	1.4	0.2	—	0.8	0.5
Golden-fronted Woodpecker	—	0.1	—	—	—	—	—
Ladder-backed Woodpecker	0.1	—	—	—	0.04	—	—
Lark Sparrow	—	0.1	0.04	—	—	—	—
Mourning Dove	0.3	—	0.4	0.1	0.2	0.07	0.1
Northern Bobwhite	0.1	—	—	0.1	—	—	0.1
Northern Cardinal	0.2	0.1	—	0.1	0.3	—	0.02
Northern Mockingbird	1.6	1.3	0.04	0.5	0.2	—	0.1
Painted Bunting	0.2	0.1	—	—	—	—	—
Red-winged Blackbird	0.04	—	—	—	—	—	—
Scissor-tailed Flycatcher	0.5	0.1	0.2	0.2	0.2	0.03	0.2
Vermilion Flycatcher	—	—	—	0.01	—	—	—
White-tailed Hawk	—	0.07	—	—	—	—	—
Yellow-billed Cuckoo	0.04	0.2	—	—	0.1	—	—

Appendix 3. Number of detections / km of all grassland bird species on combined treatment [CT], summer fire treatment [SF], Vidaurri control [VC], and Aransas control [AC] during three consecutive winter (Nov 15 to Feb 28) seasons 2005-2006, 2006-2007, and 2007-2008.

	Number of detections / km									
Species	First season			Second season			Third season			
	CT	SF	VC	CT	SF	VC	CT	SF	VC	AC
American Kestrel	0.1	0.1	—	0.3	0.2	—	0.3	—	0.3	—
American Pipit	—	0.1	—	—	—	—	—	—	—	—
American Robin	—	4.6	—	0.5	1.2	—	—	—	—	—
Ash-throated Flycatcher	—	0.1	—	0.1	—	—	—	—	—	—
Bewick's Wren	—	0.05	—	—	—	—	—	—	—	—
Blue gray Gnatcatcher	—	0.3	—	0.1	0.2	—	—	0.6	—	—
Burrowing Owl	—	—	—	0.1	—	—	—	—	—	—
Crested Caracara	—	—	—	—	0.05	—	—	—	—	—
Carolina Chickadee	—	0.3	—	—	—	—	—	—	—	—
Cassin's Sparrow	—	0.1	—	—	—	—	—	—	—	—
Clay-colored Sparrow	0.2	0.2	—	0.1	0.3	—	—	—	—	—
Eastern Bluebird	—	0.04	—	—	—	—	—	—	—	—
Eastern Meadowlark	3.9	9.4	4.2	5.8	3.5	4.0	1.3	1.4	0.9	1.9
Eastern Phoebe	0.7	1.3	—	1.2	0.8	—	0.3	0.5	—	—
Field Sparrow	—-	0.1	—	—	—	—	—	—	—	—
Golden-fronted Woodpecker	0.1	0.2	—	0.1	0.2	—	0.1	—	—	—
Grasshopper Sparrow	0.3	1.1	—	4.1	4.6	0.9	—	—	—	—
House Wren	—	1.0	—	0.6	0.9	—	—	0.8	—	—
Inca Dove	—	—	—	—	—	—	—	0.2	—	—
LeConte's Sparrow	—	0.1	—	0.1	0.3	—	—	0.2	—	—
Lincoln's Sparrow	—	1.6	—	0.1	0.8	—	—	—	—	—
Loggerhead Shrike	0.3	0.1	0.1	1.0	0.7	0.1	1.0	—	0.1	0.2
Mourning Dove	2.9	0.2	1.4	0.9	0.05	9.0	0.3	0.5	—	—
Northern Bobwhite	2.0	2.2	—	1.0	1.4	—	—	—	—	—
Northern Cardinal	0.4	0.6	—	0.6	0.4	—	—	0.2	—	—
Northern Mockingbird	0.1	0.3	—	0.6	—	—	0.1	0.2	—	—
Red-shouldered Hawk	—	—	—	—	0.05	—	—	—	—	—
Red-tailed Hawk	—	0.05	—	0.1	—	—	—	—	—	—
Ruby-crowned Kinglet	—	0.9	—	0.2	0.4	—	0.1	0.2	—	—
Sage Sparrow	27.6	10.2	16.4	12.3	7.5	6.9	1.7	2.7	0.7	1.1
Sedge Wren	0.4	0.7	0.1	1.3	0.7	0.1	1.9	1.3	1.0	4.7
Sprague's Pipit	1.0	0.3	1.8	1.1	—	1.6	—	—	—	—
Swamp Sparrow	—	—	—	1.5	1.2	—	0.5	0.8	—	—
Vesper Sparrow	0.5	0.1	—	0.3	0.4	—	—	—	—	—
White-tailed Hawk	—	0.05	0.1	—	0.05	—	0.1	—	—	—
White-tailed Kite	—	0.05	—	—	0.05	0.1	—	—	—	0.1

GREAT EGRET AND REDDISH EGRET FORAGING INTERACTIONS

Carter Garrison Crouch[1,2] and Lianne Michelle Koczur[1]

[1]*Caesar Kleberg Wildlife Research Institute, Texas A&M University-Kingsville,
Kingsville, TX 78363, USA.*

ABSTRACT.—Great Egrets (*Ardea alba*) and Reddish Egrets (*Egretta rufescens*) commonly forage in association with other wading birds. Foraging interactions are well documented in the literature for Great Egrets, yet interactions between Reddish Egrets and their foraging associates are not. Here we describe an observation of foraging interactions between Great Egrets and Reddish Egrets in southern Texas. We also discuss commensalism and the relevance of commensalism to this observation. Based on our observation we argue that some relationships that superficially appear to be commensal may be more complex. Although Great Egrets appear to benefit from foraging interactions with Reddish Egrets, Reddish Egrets may accrue costs. These costs, while likely minimal, indicate that the interactions observed between Reddish Egrets and Great Egrets may represent exploitation and not commensalism.

The Great Egret (*Ardea alba*) is an adaptable generalist with a global distribution (McCrimmon et al. 2011). Great Egrets use a vast array of habitat types for foraging including freshwater, estuarine, and marine wetlands, as well as upland habitats (McCrimmon et al. 2011). Great Egrets primarily forage using the stand-and-wait strategy or by walking slowly; however, they have been shown to employ a variety of foraging behaviors (Kushlan 1976, Kelley et al. 2003, McCrimmon et al. 2011). Great Egrets have been found to forage in association with mammals such as western lowland gorilla (*Gorilla gorilla gorilla*), forest elephant (*Loxodonta cyclotis*), forest buffalo (*Syncerus caffer nanus*), and sitatunga (*Tragelaphus spekii*) in the northern Congo (Ruggiero and Eves 1998) and with black-tailed deer (*Odocoileus hemionus*) in California (Herring and Herring 2007). They also commonly forage in mixed species assemblages with other wading birds, (Willard 1977, Kilham 1980, Caldwell 1981) as well as intraspecific flocks (Erwin 1983, Master 1992).

Reddish Egrets (*Egretta rufescens*) have a relatively restricted distribution, occurring in the United States along the southeastern Atlantic coast and the Gulf of Mexico, in Mexico along the Gulf and Pacific coasts, as well as the Sea of Cortez, and along islands in the Caribbean (Lowther and Paul 1991). Reddish Egrets are foraging habitat specialists and rely on the shallow waters of coastal lagoons to forage for small fish. Reddish Egrets often employ several active foraging techniques such as running, hopping, flying, wing-flicking and foot-stirring (Lowther and Paul 1991, Bates and Ballard 2014). They forage both solitarily and in groups with other Reddish Egrets and other heron species (Paul 1991).

In a commensal relationship one species benefits and the other suffers no negative impact (Wiens 1989). Commensalism is commonly reported in the literature, especially among avian species. Avian species exhibit commensal foraging relationships with other avian species (Baker 1980), mammalian species (Scott and Powell 1982, Levey 1999, Komar and Hanks 2002), fish (Kajiura et al. 2009, Ubaid 2011), and insects (Sutton 1951, Willis and Oniki 1978). Here we describe a foraging behavior and interaction between Reddish Egrets and Great Egrets not previously reported in the literature and discuss this interaction in terms of commensal foraging.

---

[2]Corresponding author: Carter G. Crouch, Email: CarterGCrouch@gmail.com

Reddish Egret (*Egretta rufescens*)

OBSERVATIONS

On 20 September 2015 we kayaked to a shallow mudflat at the confluence of a small creek and the Laguna Salada section of Baffin Bay in Kleberg County, Texas (27° 16’ 37” N, 97° 44’ 27” W). We remained in this area from 0930 to 1220 CST. It was a calm, sunny morning. Approximate numbers of species observed in the area included 10 Reddish Egrets (both dark and white morphs), 10 Great Egrets, two Great Blue Herons (*A. herodias*), two Snowy

Egrets (*E. thula*), one Tricolored Heron (*E. tricolor*), seven Brown Pelicans (*Pelecanus occidentalis*), 75 Laughing Gulls (*Leucophaeus atricilla*), and 15 terns (*Sternidae* spp). We observed species and foraging interactions both with the naked eye and with the aid of 12 X 50 binoculars (Leopold®).

We observed a dark morph Reddish Egret foraging, and as it became highly active in pursuing prey, four Great Egrets flew over and landed in the immediate vicinity of the Reddish Egret. Approximately 30 mins later, we noticed the behavior again. The Great Egrets stood and appeared to watch one of the Reddish Egrets at a time. Once the Reddish Egrets started actively chasing prey by running and hopping, the Great Egrets flew over and landed within 1–5 m from the actively foraging Reddish Egret. The Great Egrets would get up and fly to actively foraging Reddish Egrets from distances of approximately 10–100 m. Upon landing the Great Egrets would take pursuit of the school of fish (similar movement to a Reddish Egret) or stand and observe the Reddish Egret chasing prey. We watched Great Egrets get up and fly to foraging Reddish Egrets ~30 times during the observation period. The number of Great Egrets flying to foraging Reddish Egrets ranged from one to four per observation. The Great Egrets appeared to be cueing in on the foraging action of the Reddish Egrets and not the fish. Foraging brown pelicans were also displacing schools of small fish, but we did not observe a Great Egret fly to a pelican or the nearby school of fish.

We observed Great Egrets catch seven fish and Reddish Egrets catch five, one of which was dropped. The first time we observed a Reddish Egret successfully catch a fish while near a Great Egret, the Great Egret chased the Reddish Egret for approximately 150 m before the Reddish Egret landed. The Great Egret landed right next to the Reddish Egret. Another Reddish Egret apparently observed this take place and also flew over to land by the Reddish Egret with the fish.

We also had two observations of a Great Blue Heron flying over with Great Egrets to foraging Reddish Egrets. This may illustrate that cueing in on actively foraging Reddish Egrets may be beneficial to more than just Great Egrets.

DISCUSSION

Previous literature has reported wide variety of similar foraging interactions, with one species dispersing prey (the "beater") and the other benefitting from that dispersal. Meyerriecks and Nellis (1967) observed a Belted Kingfisher (*Megaceryle alcyon*) fly to an actively foraging Snowy Egret and successfully catch prey. Scissor-tailed Flycatchers (*Tyrannus forficatus*) have been found to follow foraging Rio Grande Turkeys (*Meleagris gallopavo intermedia*) (Baker 1980) and Ladder-backed Woodpeckers (*Dryobates scalaris*) (Kasper 2014) and successfully capture flushed insect prey. Reddish Egrets have been observed following Double-crested Cormorants (*Phalacrocorax auritus*) as they flush fish (G. Kent, pers. comm.). Kushlan (1978) found Little Blue Herons (*E. caerulea*) that foraged near White Ibis (*Eudocimus albus*) had twice as many catches per minute than herons foraging alone. Courser and Dinsmore (1975) observed Snowy Egrets flying to foraging White Ibises and following the ibises, consuming prey that was disturbed by their foraging activity. The authors also noted that this behavior was only observed when the two species were foraging on exposed mud flats or in adjacent shallow water, and not in other habitats. We observed foraging egrets on a shallow mudflat, which may have concentrated prey in the area. Reddish Egrets have been found to forage in large mixed-species flocks when prey is highly abundant (Paul 1991).

Our observations appear similar to many of the above observations, with the Reddish Egret serving as the "beater." This feeding interaction appears to be beneficial for the Great Egrets. The Great Egrets expend energy after an actively foraging Reddish Egret cues them in on the location of prey. They may also be using the Reddish Egret's active foraging technique to their advantage by landing in front of a foraging Reddish Egret. However, unlike many of the foraging associations mentioned in the literature, we hypothesize the "beaters" in this example incur a cost. Oftentimes, the Reddish Egrets would continue foraging normally, even though surrounded by Great Egrets. However, other times the Great Egrets appeared to impede foraging by the Reddish Egrets causing them to either slow or stop pursuit of prey. If there is a cost associated with a foraging association for one of the species, than it may be an example of exploitation rather than commensalism (Dodds 2009). We did not record strike rates or capture success for Reddish Egrets or Great Egrets during this observation. Strike rates and success have been documented for Great Egrets (Wiggins 1991) and Reddish Egrets (Rodgers 1983, Green 2005, Bates and Ballard 2014) foraging solitarily and in intraspecific groups; however, determining strike rates and success of both species while foraging in the manner observed would be necessary to fully understand if this foraging association is commensal or exploitative.

In studying ecological associations, researchers often categorize and label the relationships; yet ecological systems are complex and may not fall in to just one category. In a description of a commensal foraging association between little blue herons and white ibis, Kushlan (1978:679) stated, “When a heron came close to an ibis’s head, the ibis often attacked with an open billed lunge.” Elephants and gorillas have been found to show aggression towards nearby foraging Great Egrets, as well (Ruggiero and Eves 1998). These aggressive interactions may be examples of costs being incurred during an otherwise commensal association. In our observation of foraging Great Egrets and Reddish Egrets, the association appeared to be exploitative; however, interactions between these two species are not well documented and warrant future studies.

ACKNOWLEDGMENTS

We thank Bart M. Ballard, Leonard A. Brennan and M. Clay Green for their constructive comments and review. This is manuscript # 16-125 of the Caesar Kleberg Wildlife Research Institute.

LITERATURE CITED

Baker, B. W. 1980. Commensal foraging of Scissor-tailed Flycatchers with Rio Grande Turkeys. The Wilson Bulletin 92:248.

Bates, E. M. and B. M. Ballard. 2014. Factors influencing behavior and success of foraging Reddish Egrets (*Egretta rufescens*). Waterbirds 37: 191-202.

Caldwell, G. S. 1981. Attraction to tropical mixed-species heron flocks: proximate mechanism and consequences. Behavioral Ecology and Sociobiology 8: 99-103.

Courser, W. D. and J. J. Dinsmore. 1975. Foraging associates of White Ibis. Auk 92: 599-601.

Dodds, W. K. 2009. Laws, theories, and patterns in ecology. University of California Press, Berkeley.

Erwin, R. M. 1983. Feeding habitats of nesting wading birds: spatial use and social influences. The Auk 100:960-070.

Green, M. C. 2005. Plumage dimorphism in the Reddish Egret: does plumage coloration influence foraging habitat use and tactics? Waterbirds 28: 519-524.

Herring, G. and H. K. Herring. 2007. Commensal feeding of Great Egrets with black-tailed deer. Western Birds 38:299-302.

Kajiura, S. M., L. J. Macesic, T. L. Meredith, K. L. Cocks and L. J. Dirk. 2009. Commensal foraging between Double-crested Cormorants and a southern stingray. Wilson Journal of Ornithology 121: 646-648.

Kasper, S. 2014. Commensal foraging by Scissor-tailed Flycatcher over a foraging Ladder-backed Woodpecker. Bulletin of the Texas Ornithological Society 47: 28-30.

Kelly, J. F., D. E. Gawlik, and D. K. Kieckbusch. 2003. An updated account of wading bird foraging behavior. Wilson Bulletin 115: 105-107.

Kilham, L. 1980. Association of Great Egret and White Ibis. Journal of Field Ornithology 51: 73-74.

Komar, O. and C. K. Hanks. 2002. Fan-tailed Warbler foraging with nine-banded armadillos. Wilson Bulletin 114:526-528.

Kushlan, J. A. 1976. Feeding behavior or North American herons. Auk 93: 86-94.

Kushlan, J. A. 1978. Commensalism in the Little Blue Heron. Auk 95: 677-681.

Levey, D. J. 1999. Foraging Ovenbird follows armadillo. Wilson Bulletin 111: 443-444.

Master, T. L. 1992. Composition, structure, and dynamics of mixed-species foraging aggregrations in a southern New Jersey salt marsh. Colonial Waterbirds 15: 66-74.

Mccrimmon, Jr., D. A., J. C. Ogden and G. T. Bancroft. 2011. Great Egret (*Ardea alba*). The Birds of North America. Number 570.

Meyerriecks, A. J., and D. W. Nellis. 1967. Egrets serving as” beaters” for Belted Kingfishers. The Wilson Bulletin 79: 236-237.

Rodgers, Jr., J. A. 1983. Foraging behavior of seven species of herons in Tampa Bay, Florida. Colonial Waterbids 6: 11-23.

Ruggiero, R. G. and H. E. Eves. 1998. Bird-mammal associations in forest openings of northern Congo (Brazzaville). African Journal of Ecology 36: 183-193.

Scott, M. D. and J. A. Powell. 1982. Commensal feeding of Little Blue Herons with manatees. Wilson Bulletin 94: 215-216.

Sutton, G. M. 1951. Birds and an ant army in southern Tamaulipas. Condor 53: 16-18.

Ubaid, F. K. 2011. Greater Anis (*Crotophaga major*) commensal foraging with freshwater fish in the Pantanal floodplain, Brazil. Wilson Journal of Ornithology 123: 171-173.

Wiens, J. A. 1989. The ecology of bird communities volume 2: processes and variations. Cambridge University Press, New York.

Wiggins, D. A. Foraging success and aggression in solitary and group-feeding Great Egrets (*Casmerodius albus*). Colonial Waterbirds. 14: 176-179.

Willard, D. E. 1977. The feeding ecology and behavior of five species of herons in southeastern New Jersey. Condor 79: 462-470.

Willis, E. O. and Y. Oniki. 1978. Birds and army ants. Annual Review of Ecology and Systematics 9: 243-263.

ECOLOGY, BEHAVIOR, AND REPRODUCTION OF INVASIVE EGYPTIAN GEESE (*ALOPOCHEN AEGYPTIACA*) IN TEXAS

Corey T. Callaghan[1,3] and Daniel M. Brooks[2]

[1]*Environmental Science Program, Florida Atlantic University, 777 Glades Road, Boca Raton, FL 33431, USA.*
[2]*Houston Museum of Natural Science, Department of Vertebrate Zoology, 5555 Hermann Park Drive, Houston, TX 77030-1799, USA*

ABSTRACT.—Information on many aspects of Egyptian Goose (*Alopochen aegyptiaca*) life history in Texas and North America is lacking. We utilized a citizen science invasive bird project in order to collect biological data on Egyptian Geese throughout Texas. Lake/Pond was the most commonly recorded habitat (69.9%) followed by golf courses (15.6%), and rivers (9.2%), and geese were on land more often than in water. Resting (27.3%) and foraging (26.0%) were the most commonly recorded behaviors and aspects of thermoregulation are provided. The diet includes grass, aquatic vegetation, and a variety of items offered by humans. They commonly occur (24%) with other waterfowl species with few cases of agonistic behavior, and one case of hybridization with a domestic duck is documented. Mean flock size was 5.1 (range: 1-53). The geese are permanent residents, with short-distance movements common throughout the year. Breeding occurs from January to July, peaking March to May. Nesting was recorded on the ground (n = 3) and in trees (n = 2), and number of goslings ranged 2–11. Information is also provided for reproductive chronology and life cycle. Ultimately, the Texas geese appear to be generalist in nature, as they are throughout their native and nonnative ranges. The potential threats this species poses throughout its introduced range warrants further investigation.

Nonnative introduced species are generally considered to have potential negative effects on the environment (Simberloff 2000, 2003, Pyšek et al. 2012, Ricciardi et al. 2013, Blackburn et al. 2014), making nonnative species a critical component of the conservation of biodiversity worldwide. In addition to potential ecological impacts, economic (Pimentel et al. 2005, Charles and Dukes 2007, Holmes et al. 2009) and societal (Bomford 2003, Banks et al. 2008) impacts deserve attention. In avian ecology, a growing number of studies are providing anecdotal or correlative evidence of threats from introduced species to native species (Baker et al. 2014). At a minimum, proper management of nonnative introduced species relies on a basic understanding of the role they play in the novel ecosystem.

The Egyptian Goose (*Alopochen aegyptiaca*; hereafter 'EGGO') is native to Africa, particularly sub-Saharan Africa (Brown et al. 1982, Maclean 1988, Davies 2005) with a native population greater than 500,000 individuals (Banks et al. 2008). In addition to its native populations, EGGO have successfully established populations in Europe (Sutherland and Alport 1991, Delaney 1993, Lensink 1999, Rehfisch et al. 2010, Gyiemsi and Lensink 2012) and are considered one of the most rapidly spreading invasive species in Europe (Gyimesi and Lensink 2012). In North America, EGGO occur regularly in Florida, Texas, and California, among other regions (Pranty and Garrett 2011, Pranty and Ponzo 2014, eBird 2016). Information on EGGO in North America is limited to status and distribution

---

[1]Email: ccallaghan2013@fau.edu

of populations in Arkansas (Smith and James 2012, Chesbro 2015) and Florida (Pranty and Ponzo 2014), as well as a first documented nesting event in Florida (Braun 2004).

In their native range, EGGO are considered a nuisance by the public (Stephen 1985, Little and Sutton 2013) due to their prevalence on golf courses. Additionally, in their native and nonnative ranges, concerns of eutrophication through excess defecation are often reported (Stephen 1985, Little and Sutton 2013, Gyimensi and Lensink 2010, Rehfisch et al. 2010). Further, the invasive populations in Europe pose ecological and economic threats which include aggression towards native species, hybridization, eutrophication, agricultural damage, and aircraft strikes (Rehfisch et al. 2010, Gyimesi and Lensink 2010). The potential threats that the EGGO pose in Texas and North America make this species an excellent candidate for further study.

In this study we use the Texas Invasive Bird Project (TIBP), a citizen-science study targeting six invasive species in Texas (Brooks 2013), to document life history aspects surrounding the EGGO in Texas. We summarize results of the citizen-science study with particular emphasis on aspects of ecology, behavior, and reproduction of the EGGO in Texas. We also compare this information with other EGGO populations and assess any potential threats the EGGO may pose to Texas's native wildlife. This is the first study to provide information on ecology and reproduction of the EGGO in North America.

METHODS

For full details on the Texas Invasive Bird Project methodology see Brooks (2013). In brief, data were collected from a questionnaire that was designed to collect pertinent information on nonnative avian species in Texas. The questionnaire contained non-competitive questions that would elicit honest answers from competitive bird watchers. The questionnaire, available at: www.hmns.org/InvasiveBirds.doc, was distributed to birders via internet list-servs, birdwatching festivals, birdwatching clubs, and word-of-mouth.

Returned questionnaires ranged from mostly blank with many unanswered questions to fully complete with detailed information. Insufficiently completed questionnaires were not included in analyses. Photographs were often sent in with questionnaires. In order to ensure accuracy of the results, the citizen science data were proofed through checking photographs and ground truthing. The data were tabularized to conduct analyses, and results represent data that were submitted from June 2008–March 2016.

The citizen science data were supplemented through detailed observation of a bonded adult pair of EGGO that produced and fledged two goslings in early 2015. Beginning in November 2010, DMB performed weekly aquatic bird counts and monitoring at McGovern Lake in Hermann Park (Houston, Harris Co., Tx). The EGGO were observed opportunistically for a little over two years (13 January 2014 - 10 February 2016). The lake is nestled within an urban park and contains two well–planted islands, a smaller west island where the EGGO nested, and a larger east island.

RESULTS

Habitat

The most commonly recorded habitat of the EGGO was a lake and/or pond setting (69.9%; Fig. 1), followed by golf courses (15.6%), and rivers (9.2%). Most situations involved an urban component such as subdivision parks or water retention ponds. EGGO were most often recorded on land (57.6%) as opposed to water (37.7%), although in many cases the EGGO were initially observed on land but went into water as the observer approached closer (Fig. 2).

Thermal Regulation

The EGGO were capable of withstanding a wide range of temperatures (4.5 Cº–35 Cº). On 12 February an EGGO in Lufkin thermo-regulated by standing on one leg on the bank of a pond with scattered snow on the ground. An EGGO from McGovern Lake (25 December) was observed roosting at dawn on the corner of a dock 1.5 m above water following a 4.5 Cº night.

Behavior

The most frequently recorded behaviors of the EGGO (Fig. 3) were resting (27.3%), foraging (26.0%), vocalizing (12.6%), and swimming (10.0%).

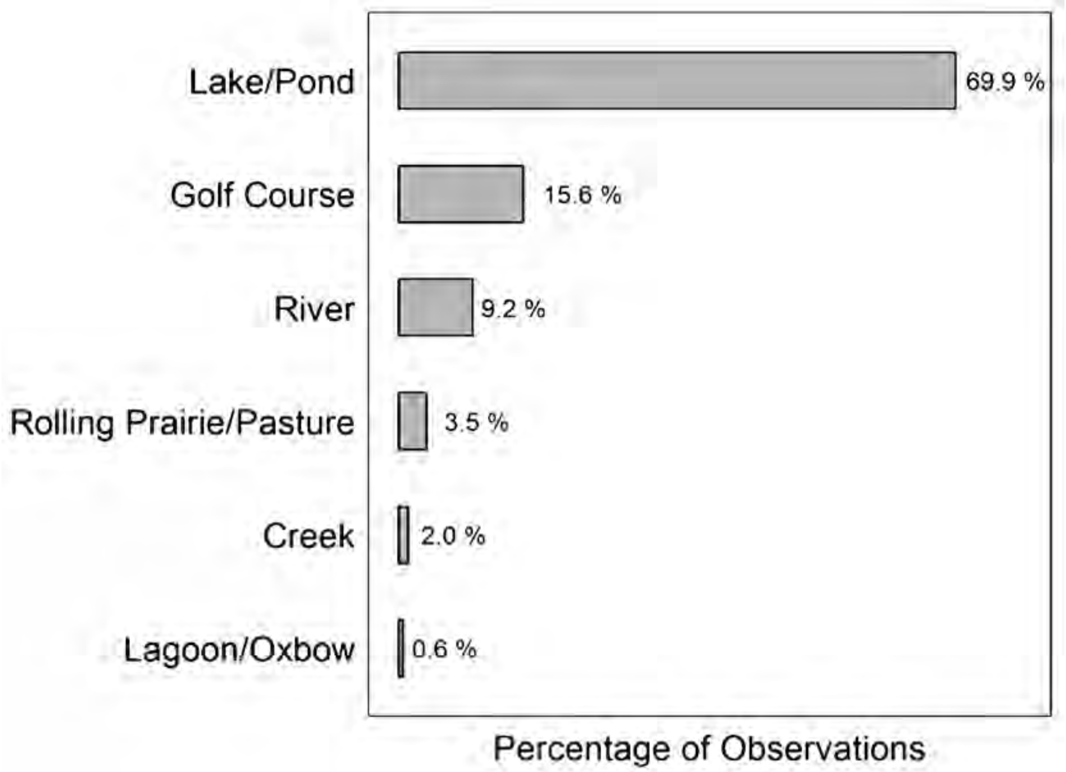


Figure 1. The percentage of habitats used by EGGO in Texas. Lake and pond were combined as the distinction was relative.

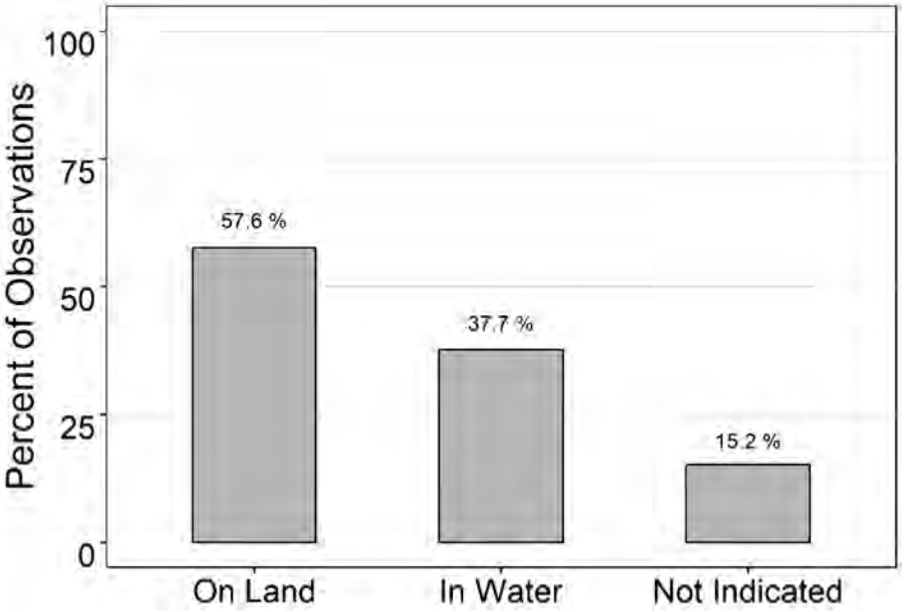


Figure 2. The percentage of EGGO in Texas using land, water, or no indication given.

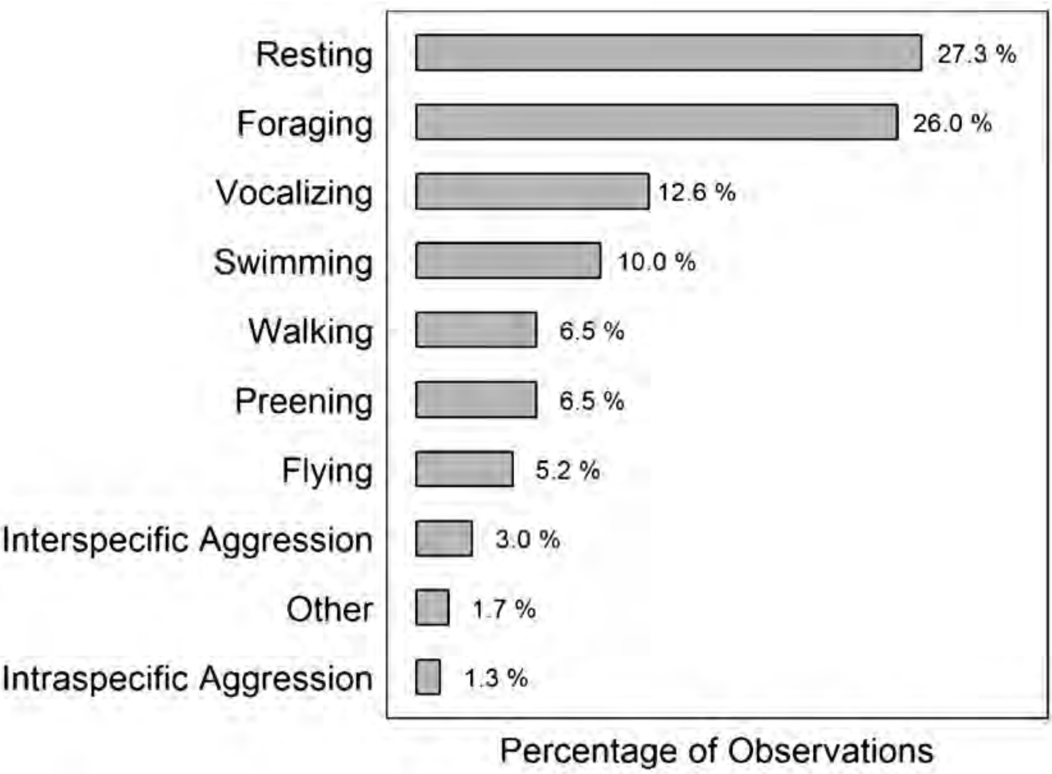


Figure 3. The most common behaviors of Egyptian Geese in Texas.

Although information on EGGO diet was rarely recorded, they utilized some type of supplemental feeding in 12% of the reports. This included: bread (n = 4), corn (n = 4), bird seed (n = 3), hen scratch (n = 2), acorns (n = 1), and dry cat food (n = 1). They were also recorded eating grass (n = 3), aquatic vegetation (n = 1), and an Almond Verbena (*Verbena virgate*) tree's spent seeds (n = 1).

EGGO were highly capable of adroit maneuverability in flight. For example, on one occasion the McGovern Lake pair parted in flight as one flew over and one under a bridge with 2 m clearance without colliding (DMB personal observation).

Interspecific interactions

EGGO were recorded with other waterfowl species 24% of the time. Commonly recorded waterfowl in association of the EGGO include: Black-bellied Whistling Duck (*Dendrocygna autumnalis*; n = 11), various domestic waterfowl species (n = 11), Muscovy Duck (*Cairina moschata domesticus*; n = 10), Mallard (*Anas platyrhynchos*; n = 8), and Ring-necked Duck (*Aythya collaris*; n = 4). Interspecific aggression was only observed on four occasions, to (n = 1 each) a Canada Goose (*Branta canadensis*), a squirrel (*Sciurus* sp.), a house cat (*Felis catus*), and a domestic duck (*A. p. domesticus*).

There was one instance of documented hybridization of an EGGO with a domestic White Pekin duck. Offspring successfully hatched and both parents aggressively protected the young. In another instance, the bonded male EGGO at McGovern Lake forced copulation with a feral female Muscovy Duck, vehemently dunking her head underwater.

Flock Dynamics

The overall mean flock size was 5.1 (mode = 2, range = 1-53, standard deviation = 7.8). The highest observed flock sizes occurred in May, June, and September, while the lowest occurred in April and August (Fig. 4). Although the most frequently recorded group size was 2, followed by solo EGGO (Fig. 5), large flocks (up to 53) were observed. There appears to be no temporal correlation of large flocks, as flocks of 10-19 were observed in March, June, September, and November; flocks of 20-49 were observed in January, June, and December; and flocks > 50 were observed in May and September.

Seasonality and movements

EGGO are non-migratory, permanent residents in Texas as they were observed throughout the calendar year (Fig. 4). Short-distant movements are typical however, as EGGO were encountered at McGovern Lake during only 45% (N = 108) of the surveys. For example, the adult pair was observed at a large concrete reflection pond ~250 m northeast of McGovern Lake on 24 August 2014, and the pair with their two grown offspring were observed at the zoo duck pond ~375 m south of the lake on 5 July 2015. The longest continuous duration the EGGO occupied McGovern Lake was during breeding (n = 18 continuous weeks).

Reproduction

In Texas, EGGO breed from January–July as goslings (2-11/brood) were recorded during these months, peaking March–May. Nesting was recorded on the ground (n = 3) and in trees (n = 2). One detailed nesting was given by an observer which documented an EGGO nest in a large Sycamore tree (*Plantanus occidentalis*) with a natural hollow at the junction of two main branches about 10 m above the ground. Both parents attended the nest, individually and simultaneously. Goslings jumped from the nest in mid-late March. A second detailed nesting event involved a parent attending a clutch of 16 eggs while being followed by a brood of 6. A third nesting event listed a man-made island in a golf course pond as the nesting location.

A brief chronology was recorded for the McGovern Lake birds. The pair appeared on 30 October 2014 after an absence of 4.5 months. Beginning 17 December 2014 only the male was seen on the south bank of the west island, serving as a sentinel, as the female incubated the clutch towards the interior of the island hidden by vegetation. The pair was first encountered with two goslings on 28 January 2015. The goslings grew quickly, attaining 50% adult size after the first month and nearly full grown at two months. The goslings dispersed from the natal site with their parents at a little more than two months of age on 1 April 2015. All four birds

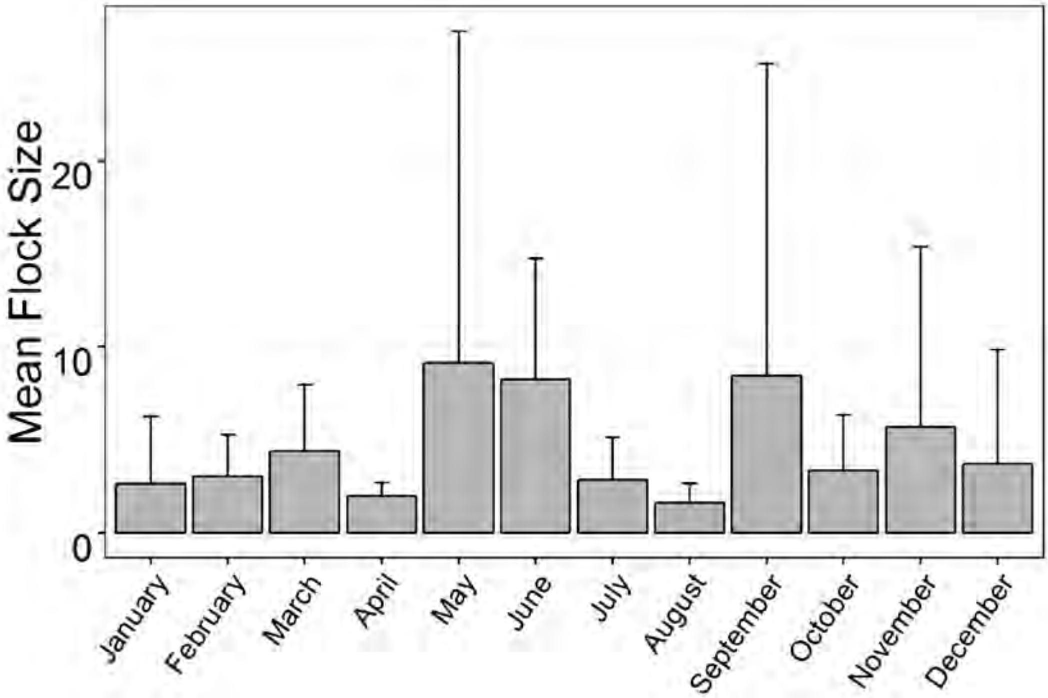


Figure 4. Mean flock size of Egyptian Geese in Texas per month, pooled across all years. Error bars represent standard deviation.

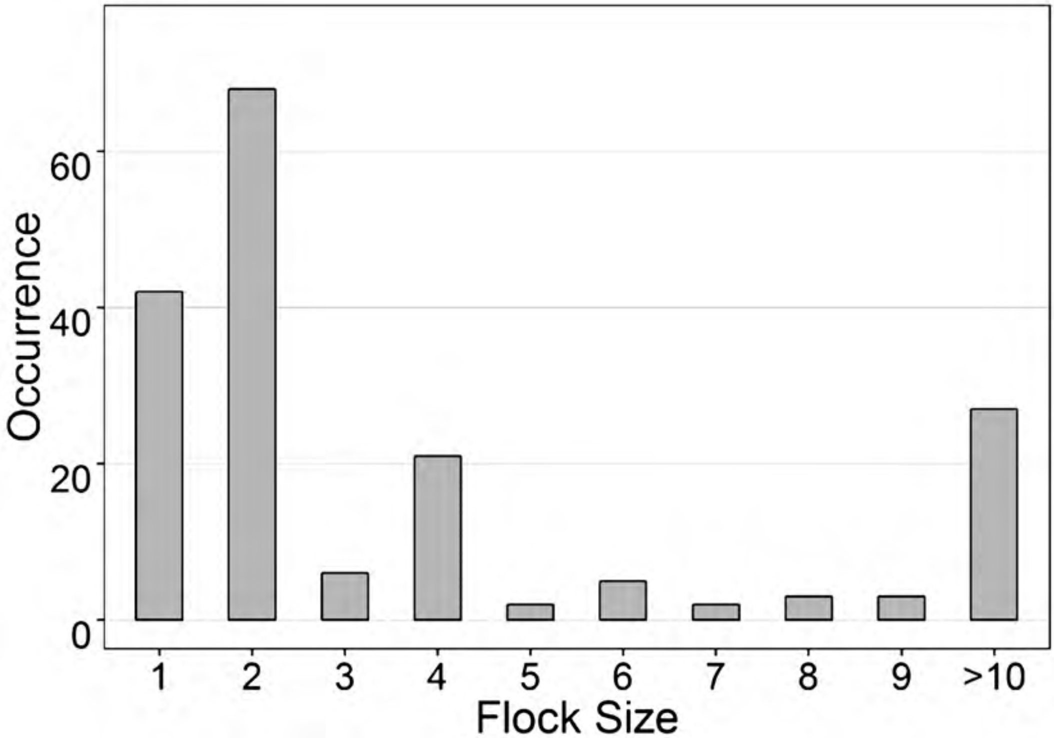


Figure 5. Frequency of flock size for Egyptian Goose in Texas.

returned once, after a three month absence, on 2 July 2015 and three days later were seen nearby on the zoo duck lake. Although the parents returned for the rest of summer and fall on 30 July 2015, the goslings were not seen again.

Predation and Mortality

There were two records of EGGO mortality by a vehicle. Another record of an unidentified species of hawk attacking goslings, which the parents defended. A photograph documented an adult EGGO predated by a Red-tailed Hawk (*Buteo jamaicensis*) as the EGGO's mate (still alive) was in the background. Raptors were present at the Herman Park site on three occasions but each time did not appear to cause distress to a single EGGO roosting alone on the corner of a dock after a chilly (4-8 Cº) evening. On two of these occasions Red-tailed Hawks were seen perching in the tree tops nearby, ~20 m from the EGGO on 31 December 2014, and ~35 m away on 8 January 2015.

DISCUSSION

EGGO in Texas spend the majority of their time resting and foraging primarily near lakes/ponds, able to persist in temperatures ranging 4.5-35 Cº. While most groups are comprised of only one or two birds, mean flock size is 5.1, but may occur in flocks exceeding 50 birds. They are permanent residents, breeding from January through July with a peak from March to May, and short local movements possible throughout the year. Currently, ecological impacts appear to be minimal, as they are amicably syntopic with a variety of other waterfowl, with agonistic behavior recorded on only four occasions to other vertebrates (n = 2 each for waterfowl and mammals), and hybridization was only documented in a single instance and reported in another.

Comparisons with Egyptian Geese in their Native Range

Throughout their native range EGGO are generalists in many aspects of their life history.

Habitat requirements are minimal as a water body is the main requisite. Rivers, lakes, ponds, marshes, reservoirs, estuaries, and pans are all utilized in their native range (Maclean 1993, Harrison et al. 1997). Diet consists of grain, crop seedlings, shoots, leaves, aquatic plants, seeds, grasses, and even insects (Brown et al. 1982, Maclean 1988, 1993). Breeding year round (Davies 2005), they are catholic in their nesting sites as they use tree cavities, cliffs, ledges, vegetation, caves, and buildings (Brown et al. 1982, Maclean 1993, Davies 2005). Predation is rarely reported, but is restricted mainly to large species of eagles (Lensink 1998), as an African Fish-Eagle (*Haliaeetus vocifer*) has been recorded pursuing an EGGO (DMB, unpublished data).

The generalist life history reported in Africa closely matches the results of this study. Habitats ranged from small subdivision ponds to large expansive lakes but generally required some water feature. Diet included grass, aquatic vegetation, Almond Verbena seeds, and a wide range of human offerings. Although nesting was rarely reported, nests were found both on the ground and in trees. EGGO are commonly found coinciding with humans and human development given that 12% of reports mentioned some form of supplemental feeding by humans. Furthermore, EGGO were commonly recorded perching on buildings, railings, docks, and other man-made structures. Lastly, while predation by Red-tailed Hawk was documented, this may be uncommon because on two different occasions a single EGGO did not seem distressed from the presence of a Red-tailed Hawk.

Throughout their native range, most potential negative effects from EGGO are of an economic nature. EGGO are considered a ‘serious pest’ (Mangnall and Crowe 2002) as their preference for cereal crops causes conflict with farmers (Maclean 1988, Mangnall and Crowe 2001, van Niekerk 2010). Further, they are considered a nuisance on many golf courses throughout South Africa (Mackay et al. 2014). Although these impacts went undocumented in our study, we highlight the potential of such conflicts given further population increase.

Comparisons with Egyptian Geese in their Invasive Range

The EGGO has successfully established populations throughout Europe (cf., Kampe-Persson 2010). Compared to the sparsely studied North American populations (Braun 2004, Smith and James 2012, Pranty and Ponzo 2014), European populations have been well studied and documented (cf., Gyimesi and Lensink 2012). The most studied EGGO population in Europe is from the Netherlands (Lensink 1996, 2002, Gyimesi and Lensink 2010, 2012). From 1967 (the first year breeding was reported) to 1999 the mean annual growth rate was 28.2% (Lensink 1996, 2002, Gyimesi and Lensink 2012). Gyimesi and Lensink (2012) most recently estimated the breeding population of EGGO in the Netherlands at 10,000 pairs, and total population at 45,000 individuals.

Sutherland and Allport (1991) report general characteristics of the species’ biology in Britain, part of its introduced range. They found EGGO had poor breeding success with an average of 1.06 young per pair. Their main habitats included parkland and rivers but in general were adjacent to water during molting. Flock size was greatest during molting where flocks of up to 50 birds were often seen. They fed on permanent grassland as well as stubble, crops, and pasture, and prefer breeding sites with short grass and open water nearby. Like in Africa, they are considered a cereal crop pest and therefore are in direct conflict with farmers in certain regions of Europe (Gyimesi and Lensink 2010).

Biology of the EGGO in Britain closely matches that reported in this study. Parkland habitat which generally included ‘ornamental lakes’ (Sutherland and Allport 1991) in Britain, is similar to parkland (i.e., golf courses, residential and suburban parks) commonly used by EGGO in Texas. Although our study documented few food items, they appear to be generalist in Texas which is similar to British populations, where they demonstrate an ability to switch from grasslands to pastures, stubble, and crops (Sutherland and Allport 1991). Flock dynamics are also similar in Texas and Britain, as the maximum flock size in our study was 53, and flock sizes of up to 50 are often seen in Britain (Sutherland and Allport 1991).

Are Invasive Egyptian Geese a Threat to the Environment in Texas?

Many potential and realized ecological, economic, and societal concerns garner attention with the feral population of EGGO in Europe (Gyimesi and Lensink 2010). These include eutrophication caused by defecation of large flocks of EGGO, defecation in public areas, fierce aggression during breeding season, and hybridization with other native species (Gyimesi and Lensink 2010, 2012).

Flock size of EGGO is a major cause for concern as flocks can consist of hundreds to thousands of birds (Gyimesi and Lensink 2010), which can lead to large defecation inputs in the environment. The highest reported observation count was 53 in our study, but this could increase given further population growth.

Curtis et al. (2007) documented a negative effect on Black Sparrowhawk (*Accipiter melanoleucus*) nest success due to EGGO. Although we did not document any negative effects on native species from EGGO, interspecific aggression occurred on only four occasions, and only two of these involved waterfowl. EGGO are known to be aggressive and territorial (Johnsguard 1978) and regarded by Beazley (1964) as the most vicious of all waterfowl when nesting. The lack of aggression observed is likely due to lack of nests found. It is also possible that there is no strain on the EGGO due to abundant resources, thereby diminishing aggression.

Hybridization of EGGO has been documented with other introduced goose and duck species (Lensink 1996, Harrop 1998, Banks et al. 2008). We received one detailed report of an EGGO hybridizing with a domestic duck. We also directly observed a male EGGO forcefully copulating with a female domestic Muscovy Duck. This was around the same time that nesting commenced. Whether this behavior was promiscuity, territoriality, or a combination of the two is unknown. Given that 24% of the time geese were reported with other waterfowl species, we acknowledge the possibility of further hybridization.

CONCLUSION

We provide a general overview of the EGGO's natural history in Texas. Given their generalist nature and abundance of habitat, nesting, and diet choices, we find it possible that the population in Texas will continue to increase and subsequently thrive. More importantly, we feel the above, combined with the ecological and economic threats the species poses throughout its natural and invaded range warrants further study. We contribute initial information on aspects of ecology, reproduction, and behavior for nonnative EGGO on which it is our fervent hopes future studies will build. Although we do not explicitly document any negative ecological or economic impacts at this time, the potential for such impacts is a cause for concern in Texas and North America.

ACKNOWLEDGMENTS

This study would not have been possible without the many individuals who provided data reports used in this study—to them we remain indebted. DMB would like to especially thank Katie Conlan for companionship in the field and help collecting data on the McGovern Lake geese. We thank Katie Conlan and Diane Harshbarger for comments that improved this manuscript.

LITERATURE CITED

Baker, J., K. J. Harvey, and K. French. 2014. Threats from introduced birds to native birds. Emu 114: 1-12.

Banks, A. N., L. J. Wright, I. M. D. Maclean, C. Hann, and M. M. Rehfisch. 2008. Review of the status of introduced non-native waterbird species in the area of the African-Eurasian Waterbird Agreement: 2007 update. British Trust for Ornithology, Norfolk, United Kingdom.

Beazley, M. 1964. The worlds atlas of birds. Mitchell Beazley Publishers Limited, Hong Kong.

Blackburn, T. M., F. Essl, T. Evans, P. E. Hulme, J. M. Jeschke, I. Kühn, S. Kumschick, Z. Marková, A. Mrugała, W. Nentwig, J. Pergl, P. Pyšek, W. Rabitsch, A. Ricciardi, D. M. Richardson, A. Sendek, M. Vilà, J. R. U. Wilson, M. Winter, P. Genovesi, and S. Bacher. 2014. A Unified Classification of Alien Species Based on the Magnitude of their Environmental Impacts. PLoS Biology 12: e1001850.

Bomford, M. 2003. Risk assessment for the import and keeping of exotic vertebrates in Australia. Bureau of Rural Sciences, Canberra, Australia.

Braun, G. D. 2004. First Documented Nesting in the Wild of Egyptian Geese in Florida. Florida Field Naturalist 32: 138-143.

Brooks, D. M. 2013. Ecology, behavior, and reproduction of an introduced population of Red-vented Bulbuls (*Pycnonotus cafer*) in Houston, Texas. Wilson Journal of Ornithology 125: 800-808.

Brown, L. H., E. K. Urban, and K. B. Newman. 1982. The birds of Africa. Academic Press, London, United Kingdom.

Charles, H. and J. S. Dukes. 2007. Impacts of invasive species on ecosystem services. Nentwig W, ed. Biological Invasions. Ecological Studies 193. Springer.

Chesbro, C. R. 2015. Distribution of the Egyptian Goose (*Alopochen aegyptiacus*) in northwestern Arkansas and in the United States of America. Thesis, University of Arkansas.

Curtis, O. E., P. A. R. Hockey, and A. Koeslag. 2007. Competition with Egyptian Geese *Alopochen aegyptiaca* overrides environmental factors in determining productivity of Black Sparrowhawks Accipiter melanoleucus. Ibis 149: 502-508.

Davies, G. 2005. Egyptian Goose *Alopochen aegyptiacus*. Pages 401-407 in Ducks, Swans, and Geese (J. Kear, Editor). Oxford University Press, Oxford, United Kingdom.

Delaney, S. 1993. Introduced and escaped geese in britain in summer 1991. British Birds 86: 591-599.

eBird. 2016. eBird: An online database of bird distribution and abundance [web application]. eBird, Ithaca, New York. http://www.ebird.org. (accessed: 7 May 2016).

Gyimesi, A. and R. Lensink. 2010. Risk analysis of the Egyptian Goose in The Netherlands. Consultants for Environment and Ecology, pp. 88.

Gyimesi, A. and R. Lensink. 2012. Egyptian Goose *Alopochen aegyptiaca*: an introduced species spreading in and from the Netherlands. Wildfowl 62: 128-145.

Harrison, J. A., D. G. Allan, L. G. Underhill, M.

Harrop, A. H. J. 1998. Successful hybridisation between Ruddy Shelduck and Egyptian Goose. British Birds 91: 281.

Herremans, A. J. Tree, V. Parker, and C. J. Brown. 1997. The atlas of southern African birds. BirdLife South Africa, Johannesburg, South Africa.

Holmes, T. P., J. E. Aukema, B. von Holle, A. Liebhold,

and E. Sills. 2009. Economic Impacts of Invasive Species in Forests: Past, Present, and Future. The Year in Ecology and Conservation Biology, 2009: Annual New York Academy of Sciences 1162: 18-38.

Johnsguard, P. A. 1978. Egyptian Goose. Pages 115-116 in Ducks, Geese, and Swans of the World. University of Nebraska Press, Lincoln.

Kampe-Persson, H. 2010. Occurrence of Egyptian Goose *Alopochen aegyptiaca* in Europe. Goose Bulletin: 34-37.

Lensink, R. 1996. Feral breeding of naturalized birds in the Netherlands; history, present day situation and future. Limosa 69: 85-103.

Lensink, R. 1998. Temporal and Spatial Expansion of the Egyptian Goose *Alopochen aegyptiacus* in the Netherlands, 1967-94. Journal of Biogeography, 25: 251-263.

Lensink, R. 1999. Aspects of the biology of Egyptian Goose *Alopochen aegyptiacus* colonizing The Netherlands. Bird Study 46: 195-204.

Lensink, R. 2002. Nijlgans. In S. V. Nederland (eds.), Atlas van de Nederlandse broedvogels 1998-2000 pp. 108-109. KNNV Uitgeverij, Leiden. [In Dutch with English summary.]

Little, R. M. and J. L. Sutton. 2013. Perceptions towards Egyptian Geese at the Steenberg Golf Estate, Cape Town, South Africa. Ostrich 84: 85-87.

Mackay, B., R. M. Little, A. Amar, and P. A. R. Hockey. 2014. Incorporating environmental considerations in managing Egyptian Geese on golf courses in South Africa. The Journal of Wildlife Management 78: 671-678.

Maclean, G. L. 1988. The Egyptian Goose *Alopochen aegyptiacus*. Journal of East Cape Game Management Association Pelea 7: 30-39.

Maclean, G. L. 1993. Robert's Birds of Southern Africa. John Voelcker Bird Book Fund, Cape Town, South Africa.

Mangnall, M. J. and T. M. Crowe. 2001. Managing Egyptian Geese on the croplands of the Agulhas Plain, Western Cape, South Africa. South African Journal of Wildlife Research 31: 25-34.

Mangnall, M. J. and T. M. Crowe. 2002. Population dynamics and the physical and financial impacts to cereal crops of the Egyptian Goose *Alopochen aegyptiacus* on the Agulhas Plain, Western Cape, South Africa. Agriculture, Ecosystems & Environment 90: 231-246.

Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52: 273-288.

Pranty, B. and K. L. Garrett. 2011. "Non-countable" Exotic Birds in the ABA Area. Birding: 46-58.

Pranty, B. and V. Ponzo. 2014. Status and distribution of Egyptian Geese (*Alopochen aegyptiaca*) in Southeastern Florida. Florida Field Naturalist 42: 91-139.

Pyšek, P., V. Jarošík, P. E. Hulme, J. Pergle, M. Hejda,

U. Schaffner, and M. Vilà. 2012. A global assessment of invasive plant impacts on resident species, communities, and ecosystems: the interaction of impact measures, invading species' traits and environment. Global Change Biology 18: 1725-1737.

Rehfisch, M. M., J. R. Allan, and G. E. Austin. 2010. The effect on the environment of Great Britain's naturalized Greater Canada Branta canadensis and Egyptian Geese *Alopochen aegyptiacus*. BOU Proceedings_The Impacts of Non-native Species 2: 1-9.

Ricciardi, A., M. F. Hoopes, M. P. Marchetti, and J. L.

Lockwood. 2013. Progress toward understanding the ecologocial impacts of non-native species. Ecological Monographs 83: 263-282.

Simberloff, D. 2000. Global climate change and introduced species in United States Forests. The Science of the Total Environment 262: 253-261.

Simberloff, D. 2003. Confronting introduced species: A form of xenophobia? Biological Invasions 5: 179-192.

Smith, K. G. and D. A. James. 2012. History and current status of Egyptian Goose (*Alopochen aegyptiacus*) in Northwestern Arkansas. Journal of the Arkansas Academy of Science 66: 200-204.

Stephen, V. 1985. A review of mitigation measures and their effectiveness in reducing the nuisance of geese on golf courses. Thesis, University of Cape Town.

Sutherland, W. J., and G. Allport. 1991. The distribution and ecology of naturalized Egyptian Geese *Alopochen aegyptiacus* in Britain. Bird Study 38: 128-134.

van Niekerk, J. H. 2010. Assemblages and movements of waterfowl at cattle feedlots across Gauteng, South Africa. Ostrich 81: 31-37.

JESSE MILLER: EGG COLLECTOR, ENGINEER AND PHILANTHROPIST

Stanley D. Casto[1,2]

Department of Biology, University of Mary Hardin-Baylor, Belton, Texas 76513

ABSTRACT.— Jesse Wright Miller (1874-1919) published six brief notes on Texas birds based on observations made during 1891 through 1893. His personal collection of eggs is now included in the Biodiversity Research and Teaching Collection at Texas A&M University. This paper reviews the life of Jesse Miller and his contributions to the ornithology of Texas.

Jesse Wright Miller (1874-1919, fig. 1), third son of Charles and Annexa Brashear Miller, was born in Houston, Texas, on 6 December 1874. His father was a wealthy planter, and his mother was the daughter of Isaac Brashear, a signer of the Texas Annexation Ordinance. Besides Jesse, there were three additional children in the Miller family: Earnest Brashear (b. 1868), Isaac Austin (b. 1869), and Alma (b. 1879). Alma Miller married Kenneth E. Womack, Sr., and it was through this family that a collection of eggs belonging to Jesse Miller was saved for posterity. This paper reviews the education and professional life of Jesse Miller, his articles on Texas birds, and his collection of eggs in the Biodiversity Research and Teaching Collection at Texas A&M University.

Figure 1. **Jesse Wright Miller** (1874-1919). Yale Class Book 1900, Manuscripts and Archives, Yale University Library.

EDUCATION AND PROFESSIONAL LIFE

Jesse Miller was educated in the public schools of Houston and at Phillips Academy in Andover, Massachusetts. After leaving Phillips, he enrolled at Yale University where he graduated in 1900 with a specialty in mining engineering. The summer following his graduation from Yale was spent in Quebec, Canada, recovering from a surgical procedure. Miller then enrolled at the University of Texas and during the academic year 1901-1902 completed a two-year curriculum in the study of law, a feat attesting to his intelligence and diligence in the pursuit of a goal.

His first job following law school was teaching English in the Philippines, a position that he left the following year to work for the company constructing the Canton-Hankow Railway in China. In 1903, Miller returned to New York State where he worked as an engineer on a water project for a few months before moving to West Virginia to work as a draftsman-topographer for the United States Coal and Coke Company. He left this employment in 1904 to become superintendent of a company that

---

[1] Present address: 159 Red Oak, Seguin, Texas.

[2] Email: sscasto2@aol.com

operated mines in the State of Guerrero, Mexico, but soon resigned to take a similar position in the State of Sonora. During 1907, Miller was in Ontario, Canada, on mining business but the following year returned to Mexico where he continued in the mining profession until 1909. He left Mexico in 1910 to examine placer mines in Colombia, South America. Then, for reasons unknown, he abandoned his career in mining and returned to United States and his alma mater, Yale University.

Miller took a Master of Arts Degree from Yale University in 1913. He then returned to Houston where from 1914 until 1918 he was a trader with the Cotton Exchange. In this capacity, he became well known and accumulated considerable wealth. Then, in a sudden career change, he left Houston for Redlands, California. On 21 June 1919, he was killed in Los Angeles, California, when a streetcar struck the automobile in which he was riding. He never married and, at the time of his death, was survived only by his younger sister, Alma Miller Womack. Miller was a member of the American Institute of Mining Engineers, and of several scientific societies and social clubs in New York, Houston, and Mexico (Barbour 1926, Whittlesey 1912, Yale University 1920).

Jesse Miller left some of his property to the City of Houston. When the city found that the property was not suitable for their purposes, it was sold to Alma Miller Womack, for the sum of $50,000. This money was then used to fund construction of an outdoor theatre in Hermann Park. This theatre, dedicated on 18 May 1923, was named the Miller Outdoor Theatre in memory of its benefactor, Jesse Wright Miller. In 1966 most of the original theater was torn down. The present-day theatre, constructed on the same site and dedicated in September 1968, is still known as the Miller Outdoor Theatre (Anon. 1923, 1968; Streckfuss 1994).

ARTICLES ON TEXAS BIRDS

Jesse Miller reportedly spent “some years of communion with nature afloat and ashore…” before he enrolled at Phillips Academy (Decrow 1900). It would thus seem that his interest in birds was acquired from influences within the home and from associates in the Houston area. He probably began to collect eggs about 1890, and in the following year placed advertisements in *The Oologist* and *The Collectors’ Monthly* expressing his desire to exchange eggs of herons, grackles, Brown-headed Cowbird, Yellow-breasted Chat, Blue-gray Gnatcatcher, Purple Martin and other species (Miller 1891a,c). Miller continued to advertise in *The Oologist* through the summer of 1894 while attending classes at Phillips Academy in Massachusetts (Miller 1894a).

Miller published six brief articles on Texas birds. The first of these notes was a letter to the editor of *The Collectors Monthly* describing the nest of a Great Crested Flycatcher made of small sticks, moss, cotton seed hulls, and lined with hair, feathers and snake skin (Miller 1891b). A second note described a trip on 7 May 1892 to DeWalt Lake in Fort Bend County, Texas, during which he was bitten by a wounded Anhinga. The nest of a Wood Duck was later found in a gum tree but the eggs were not obtained because the tree could not be climbed. Swallow-tailed Kites were seen over the lake “sweeping and skimming the water like martins and swallows,” a behavior correctly surmised to be their way of drinking. One of the kites was seen carrying Spanish moss to its nest in a large cottonwood (Miller 1894b).

The third article described a nesting colony of Great-tailed Grackles at a pond near Mason Lake in Fort Bend County on 15 May 1892. Seventy-five to 100 nests were placed 2-3 feet above the water and about the same distance from the tops of the “saw grass” to which they were attached. Contrary to present-day observations, Miller noted that grackles in Harris and Fort Bend counties seemed to “prefer isolated, rather than settled portions of the country” and that they were “rarely seen about towns.” The nest of a Least Bittern with 5 eggs was also found in the grackle colony. Based on this observation and a similar finding a few years earlier, Miller speculated that the Least Bittern nested in association with the Great-tailed Grackle because of the protection afforded by the stronger and more pugnacious grackles (Miller 1895).

Miller’s most significant find was made on 11 March 1893 while on an outing near Houston. A small bird carrying nesting material was seen entering a hole in the dead snag of a gum tree. The bird soon left the hole and since it did not return within a reasonable time, Miller continued deeper into the forest. He returned to the gum tree later in the day where he again saw the bird enter the hole while carrying an insect. Climbing to the entrance of the nest, Miller held onto the snag with one hand while using his pocketknife to cut through the rotten wood into the nest cavity. Five well-feathered nestlings were found in the nest. The parent birds that perched nearby during the removal of the young were identified as Brown-headed Nuthatches (*Sitta pusilla*) (Miller 1894c). Although Henry Nehrling recorded the Brown-headed Nuthatch in Lee County in 1882, the nest taken by Jesse Miller represents the first documentation of the species nesting in Texas (Oberholser n.d.:7038-39).

A short note published in the April 1893 issue of *The California Traveller and Naturalist* described the difficulty of hunting Wilson's snipe around Houston. Miller noted that snipe were found along the central coast from November through March with the latest bird being seen on April 18th. The belief was also expressed that the Texas coast was not the southernmost point of migration for Wilson Snipe since in excessively cold winters they usually came in greater numbers than usual (Miller 1893).

Miller's last paper was based on observations made on 10 May 1893 while he was walking along a cow path over "an open prairie" near Houston. A Grasshopper Sparrow (*Ammodramus savannarum*) exhibiting distress behavior suddenly flushed from the grass beside the trail. A close examination revealed four eggs in a partly roofed nest built in a slight depression. The roof of the nest covered "nearly two-thirds of the nest, and…was so placed that the settling bird could be shaded for most of the day." Although abandoned nests of this type had been found on previous occasions, it was not until this time that Miller knew the identity of the birds that had built them (Miller 1894d).

Jesse Miller was reportedly the author of a number of magazine articles (Yale University 1920). The nature of these articles is unknown but it assumed that they dealt with topics in mining engineering and cotton trading. The publication of his bird notes in ephemeral journals such as The Naturalist [Austin, TX], *The California Traveller and Naturalist* [San Jose, CA], *The Oregon Naturalist* [Portland, OR] and *The Collectors' Monthly* [Danielsonville, CT] effectively ensured that his contributions to ornithology would quickly be forgotten. No record has been found that Miller was ever a member of any ornithological society.

EGG COLLECTION

In 1991, Kenneth E. Womack, Jr., son of Alma Miller Womack, donated a collection of eggs to the Texas Wildlife Cooperative Collection at Texas A&M University [now the Biodiversity Research and Teaching Collection]. These eggs were believed by their donor to have been collected or acquired through trading by Isaac Austin Miller during the period from 1900 to 1910, and that their owner was the person for whom the Miller Outdoor Theatre in Houston is named (Arnold 1991).

It now seems that the original owner of the eggs was misidentified. There is no evidence that Isaac Austin Miller (1869-1916) was an egg collector, and he is not the person for whom the Miller Outdoor Theatre is named. Austin Miller did, however, occupy a position of prominence in Houston where for many years he was the city engineer and county surveyor (Anon. 1916). Based on what is now known, it is reasonable to assume that the eggs donated in 1991 are a remnant of those belonging to Jesse Miller and that they were most likely acquired during the period from 1890 to 1900.

Many of the eggs in the Miller Collection have no data, but some boxes do contain small slips on which are written measurements of the eggs, the set designation (e.g. 1/5, one set with 5 eggs), and an identifying checklist number or numbers. In a few cases, a generic or binomial name is written on the slips. There are no dates or locations given for any of the eggs nor is Jesse Miller's name or initials found on any of the eggs or their data slips.

The use of the identifying checklist numbers is inconsistent. Some of the numbers are those used by the American Ornithologists' Union whereas others seem to be numbers used in the Baird, Coues, and Ridgway checklists. In some cases, there are two identifying numbers given for a single set of eggs.

It is unknown how many species of birds are represented in the collection. Several of the sets seem to be duplicates. All of the eggs appear to be from birds found in the United States and Canada. Eggs identified on the data slips in the collection include those of the Montezuma Quail, Common Murre, Thick-billed Murre, Tufted Puffin, Razorbill, Horned Lark, Bushtit, Cliff Swallow, Barn Swallow, Tufted Titmouse, House Wren, Sedge Wren, Cedar Waxwing, House Finch, Yellow Warbler, and House Sparrow.

Only a few of the eggs collected by Jesse Miller are in museums or recorded in the literature. The Western Foundation of Vertebrate Zoology has eggs of Yellow-crowned Night Heron and Snowy Egret collected in Fort Bend County on 17 May 1891 (Corado 2005). Single egg sets of a Carolina Chickadee taken in Harris County on 5 May 1891 and of a Snowy Egret taken on 10 May 1891 at Mason Lake in Fort Bend County are in the National Museum of Natural History. Egg sets of the Carolina Chickadee, Common Grackle, Brown-headed Cowbird, and Snowy Egret are in the California Academy of Sciences Museum of Natural History. The typescript of *The Bird Life of Texas* also cites a set of Snowy Egret eggs taken at Mason Lake, Fort Bend County, on 8 July 1894 (Oberholser n.d.:1359).

ACKNOWLEDGMENTS

I thank Keith Arnold for drawing my attention to the eggs collected by Miller and providing access to the collection. Copies of Jesse Miller's obscure publications were obtained through the efforts of Shiloh Fulton. This study was supported in part by a Wells Research Professor grant from the University of Mary Hardin-Baylor.

LITERATURE CITED

ANON. 1916. I. Austin Miller [obituary]. Houston Daily Post, 19 January 1916.

_______. 1923. Fair audience attends play at Miller Theater. Houston Chronicle, 20 May 1923.

_______. 1968. Jesse Wright Miller. Houston Chronicle, 25 August 1968.

ARNOLD, K. A. 1991. Letter from K. A. Arnold to the author, 18 October 1991.

BARBOUR, L. B. 1926. Quarter-century record of the class of 1900 Yale College. New Haven: Class Secretaries Bureau.

CORADO, R. 2005. E-mail from René Corado, Western Foundation of Vertebrate Zoology, to the author dated 29 April 2005.

DECROW, J. W. 1900. Yale class book 1900. New Haven: Yale University.

MILLER, J. W. 1891a. To exchange. The Oologist 8(9): [advertisement section].

_______. 1891b. Letter to the editor. The Collectors' Monthly 2(8):43.

_______. 1891c. To exchange. The Collectors' Monthly 2(9): [advertisement section].

_______.1893. Wilson's snipe. The California Traveller and Naturalist 2(7): [no pagination].

_______. 1894a. Exchange column. The Oologist 11(9):[advertisement section].

_______. 1894b. Notes on wood duck, water turkey, and swallow-tailed kite. The Naturalist 1(3):32-33.

_______. 1894c. Nesting of the brown-headed nuthatch in Texas. The Naturalist 1(4):36-38.

_______. 1894d. Nesting of the grasshopper sparrow in Texas. The Naturalist 1(5):54-56.

_______. 1895. Nesting of the least bittern and Texas grackle. The Oregon Naturalist 2(2):25.

OBERHOLSER, H. C. [n.d.]. Typescript of The Bird Life of Texas (microfilm). Briscoe Center for American History, University of Texas at Austin.

STRECKFUSS, A. L. 1994. The Miller Outdoor Theatre. Houston: Privately published.

WHITTLESEY, G. N. 1912. Decennial record of the class of 1900 Yale College. New Haven:Tuttle, Moorehouse & Taylor Company.

YALE UNIVERSITY. 1920. Obituary record of graduates deceased during the year ending July 1, 1919. New Haven: Yale University.

TEXAS BIRD RECORDS COMMITTEE REPORT FOR 2015

Eric Carpenter[1]

4710 Canyonwood Drive, Austin, Texas 78735

The Texas Bird Records Committee (hereafter "TBRC" or "committee") of the Texas Ornithological Society requests and reviews documentation on any record of a TBRC Review List species (see TBRC web page at http://www.texasbirdrecordscommittee.org). Annual reports of the committee's activities have appeared in the Bulletin of the Texas Ornithological Society since 1984. For more information about the Texas Ornithological Society or the TBRC, please visit www.texasbirds.org. The committee reached a final decision on 58 records during 2015: 49 records of 30 species were accepted and 8 records of 7 species were not accepted, an acceptance rate of 84.48% for this report. A total of 102 observers submitted documentation (to the TBRC or to other entities) that was reviewed by the committee during 2015.

The TBRC accepted three first state records in 2015. The additions of Bar-tailed Godwit, Red-legged Honeycreeper and Gray-crowned Rosy-Finch bring the official Texas State List to 642 species in good standing. This total does not include the five species on the Presumptive Species List.

In addition to the review of previously undocumented species, any committee member may request that a record of any species be reviewed. The committee requests written descriptions as well as photographs, video, and audio recordings if available. Information concerning a Review List species may be submitted to the committee secretary, Eric Carpenter, 4710 Canyonwood Drive, Austin, Texas 78735 (email: ecarpe@gmail.com). Guidelines for preparing rare bird documentation can be found in Dittmann and Lasley (1992) or at *http://www.greglasley.net/document.html*.

The records in this report are arranged taxonomically following the AOU Check-list of North American Birds (AOU 1998) through the 56th supplement (Chesser et al. 2015). A number in parentheses after the species name represents the total number of accepted records in Texas for that species at the end of 2015. Species added to the Review List because of population declines or dwindling occurrence in recent years do not have the total number of accepted records denoted as there are many documented records that were not subjected to review (e.g. Brown Jay, Pinyon Jay, Tamaulipas Crow, and Evening Grosbeak). All observers who submitted written documentation or photographs/recordings of accepted records are acknowledged by initials. If known, the initials of those who discovered a particular bird are in boldface but only if the discoverer(s) submitted supporting documentation. The TBRC file number of each accepted record will follow the observers' initials. If photographs or video recordings are on file with the TBRC, the Texas Photo Record File (TPRF) (Texas A&M University) number is also given. If an audio recording of the bird is on file with the TBRC, the Texas Bird Sounds Library (TBSL) (Sam Houston State University) number is also given. Specimen records are denoted with an asterisk (*) followed by the institution where the specimen is housed and the catalog number. The information in each account is usually based on the information provided in the original submitted documentation; however, in some cases this information has been supplemented with a full range of dates the bird was present if that information was made available to the TBRC. All locations in italics are counties. Please note that the county designations of offshore records are used only as a reference to the nearest point of land.

TBRC Membership—Members of the TBRC during 2015 who participated in decisions listed in this report were: Randy Pinkston, Chair; Keith Arnold, Academician; Eric Carpenter, (non-voting) Secretary; Greg Cook, Tim Fennell, Mary Gustafson, Petra Hockey, Mark Lockwood, Jim Paton, Byron Stone, Dan Jones, Stephan Lorenz. During 2015, Mary Gustafson's and Tim Fennell's second term each expired with Dan Jones and Stephan Lorenz elected to fill those vacancies. The

---

[1] Email: ecarpe@gmail.com

Chair, Academician, and Secretary were also re-elected.

Contributors—Lindsey Augustus, Eddie Bartley (**EBa**), Mark Bartosik (**MBa**), Aaron Baxter, Theresa Bayoud, Mikael Behrens (**MBe**), Bob Behrstock, Chris Benesh, David Brotherton, Luanne Brotherton, Erik Bruhnke (**EBr**), Kelly Bryan, Kirby Campbell, Eric Carpenter, Larry Carpenter, Bala Chennupati, Sheridan Coffey (**ShC**), Scarlet Colley (**ScC**), Steve Collins (**StC**), Mel Cooksey, Tim Cooper, Rich Damron, Tripp Davenport, Kristin Davis, Carolyn Dill, Gil Eckrich, Mark Esparza, Ed Fair, Sam Fason, Robert Ferrell, Joe Fischer, Phyllis Frank, Tony Frank, Bret Fried, Pam Goolsby, Lisa Griffis, Saul Grysman, Mary Gustafson, John Haynes, Mitch Heindel, Petra Hockey, Scott Holt, Eric Hough, Dan Jones, John Kaye, Tiffany Kersten, Florence King, Rich Kostecke, Mark Kulstad, Greg Lasley (**GLs**), Greg Lavaty (**GLv**), Cin-Ty Lee (**CTL**), Natalie Lindholm, Mark Lockwood, Jud Maxwell (**JMa**), Donna McCown (**DMc**), Steven McDonald (**StM**), Sean McElaney (**SeM**), Nate McGowan, Craig McIntyre (CrM), Jon McIntyre (**JMc**), Brad McKinney, Candy McNamee (**CaM**), Arman Moreno (**AMo**), Alisa Muniz (**AMu**), Derek Muschalek (DMu), Bob Neiman, Patrick Nowlin, John O'Brien, Dale Ohl, Andrew Orgill, Brent Ortego, Sue Orwig, Greg Page (**GPa**), Jim Paton (**JPa**), Wendy Petty, Barrett Pierce, Randy Pinkston, Sharon Pratt, Gene Prejean (**GPr**), Joann Pruitt (**JPr**), Jacob Roalef (**JaR**), Dan Roberts, John Rosford (**JoR**), Chris Runk, Suzanne Schroeder, Willie Sekula, Paul Sellin (**PSe**), Chris Seymour, Dennis Shepler, Rhoda Spuhler (**RSp**), Eric Stager, Rex Stanford (**RSt**), Jim Stevenson, Byron Stone, Paul Sunby (**PSu**), Samuel Taylor, Margaret Viens, Fred Welden, Teri Zambon, Barry Zimmer, Robin Zurovec.

Acknowledgments—The TBRC is very grateful to the many contributors listed above, without whom this report would not be possible. The committee would also like to thank Martha Jordon, Steve Mlodinow, Nathan Pieplow and Peter Pyle for providing the TBRC with expert opinion concerning records reviewed during 2015. The author thanks Jim Peterson and TBRC members for reviewing previous drafts of this report.

Additional Abbreviations—AOU = American Ornithologists' Union; NP = National Park; NS = National Seashore; NWR = National Wildlife Refuge; SHS = State Historic Site; SNA = State Natural Area; SP = State Park; TBSL = Texas Bird Sounds Library (Sam Houston State University); TCWC = Texas Cooperative Wildlife Collection (Texas A&M University); WMA = Wildlife Management Area.

ACCEPTED RECORDS

Trumpeter Swan (*Cygnus buccinator*) (12). One at The Woodlands, Montgomery, from 9 - 22 December 2014 (**GPr**, LC, LG, FK, PSe, TF, PF, RP, BS; 2014-66; TPRF 3225).

Eurasian Wigeon (*Anas penelope*) (54). One se. of Cedar Park, Williamson, on 18 January 2015 (**MBe**, RK; 2015-06; TPRF 3226).

Red-necked Grebe (*Podiceps grisegena*) (29). One at Red Bluff Lake, Reeves/Loving, on 20 March 2015 (**BN**; 2015-24; TPRF 3233).

Red-billed Tropicbird (*Phaethon aethereus*) (14). One off South Padre Island, Cameron, on 25 October 2014 (**EC**, **AMo**, **RP**, **PH**, **BM**; 2014-56; TPRF 3220).

Jabiru (*Jabiru mycteria*) (11). One at a private ranch, Victoria, on 6 August 2014 (**BO**; 2014-43; TPRF 3213).

Brown Booby (*Sula leucogaster*) (48). As many as five at Corpus Christi & Nueces Bays, Nueces/San Patricio, from 15 December 2013–30 July 2014 (JMc, CaM, CrM, TZ, DR, RSt, PG, SH, AB; 2013-79; TPRF 3201). One at Espiritu Santo Bay, Calhoun, on 17 June 2014 (**PH**; 2014-35; TPRF 3207). One at South Padre Island & Port Isabel, Cameron, from 28 June–12 July 2014 (**ScC**, RZ, SS; 2014-39; TPRF 3210). Up to two offshore, SPI Pelagic, Cameron, on 16 August 2014 (**EC**, **PH**, **TD**, **GLv**; 2014-45; TPRF 3215). One at Galveston Ship Channel, Galveston, on 24 August 2014 (**CTL**, **MK**; 2014-47). One at Matagorda Bay, Calhoun/Matagorda, on 27 August 2014 (**PH**; 2014-46). One at Windy Point, Lake Travis, Travis, from 6 September 2014–8 January 2015 (**RK**, EC, AMo, EF, RP, JaR; 2014-48; TPRF 3216). One at Matagorda jetties, Matagorda, on 15 December 2014 (**PH**; 2014-68). One at Pleasure Island, Jefferson, on 17 February 2015 (**PN**; 2015-16; TPRF 3230).

Northern Goshawk (*Accipiter gentilis*) (25). One at Utopia, Uvalde, on 19 March 2014 (**MH**; 2014-40).

Collared Plover (*Charadrius collaris*) (2). One at Hargill playa, Hidalgo, from 2–17 August 2014

(**DJ**, RD, NM, ShC, EC, RP, PF, TF, DS, PH, RSt, MC, BS; 2014-41; TPRF 3211).

Bar-tailed Godwit (*Limosa lapponica*) (1). One at Oso Bay, Nueces, from 9 August–26 October 2014 (**MC**, EC, PF, TF, BM, RSt, RP, SF, PH, TB, StM, AO; 2014-44; TPRF 3214). This represents the first documented record for Texas.

Red Phalarope (*Phalaropus fulicarius*) (45). One at Marathon sewage ponds, Brewster, from 15–16 June 2014 (**DO**, ML; 2014-34; TPRF 3077).

Long-tailed Jaeger (*Stercorarius longicaudus*) (25). One at Lake Belton, Bell, on 8 September 2014 (**GE**; 2014-50; TPRF 3217). One off South Padre Island, Cameron, on 20 September 2014 (**AMo**, **EBr**, **BM**; 2014-52; TPRF 3218).

Black-legged Kittiwake (*Rissa tridactyla*) (99). One at Texas City Dike, Galveston, on 14 December 2014 (**MBa**; 2014-70; TPRF 3238).

Mew Gull (*Larus canus*) (38). One at Keystone Heritage Park, El Paso, El Paso, on 28 December 2014 (**JPa**; 2015-15; TPRF 3229).

Elegant Tern (*Thalasseus elegans*) (6). One at Texas City Dike, Galveston, on 1 November 2014 (**MBa**; 2014-72; TPRF 3239).

Green Violetear (*Colibri thalassinus*) (77). One at nw. Austin, Travis, from 27 - 29 May 2014 (**JMa**; 2014-32; TPRF 3204). One at La Vernia, Wilson, from 27 June - 1 July 2014 (**RF**, ShC, NM; 2014-38; TPRF 3209).

White-eared Hummingbird (*Hylocharis leucotis*) (35). Up to three at Davis Mountains Resort, Jeff Davis, from 25 May–4 September 2014 (**KB**, ML; 2014-37; TPRF 3208).

Sulphur-bellied Flycatcher (*Myiodynastes luteiventris*) (26). One at Paradise Pond, Port Aransas, Nueces, on 22 May 2011 (**CS**; 2014-42; TPRF 3212). One at w. Galveston Island, Galveston, from 5–13 May 2014 (**JS**, RSp; 2014-33; TPRF 3205).

Fork-tailed Flycatcher (*Tyrannus savana*) (29). One at Sabine Woods, Jefferson, on 22 September 2014 (**JH**; 2014-54). One at Laguna Atascosa NWR, Cameron, from 27–28 November 2014 (**BC**, SP, ME; 2014-62; TPRF 3222). One at w. Galveston Island, Galveston, on 22 December 2014 (**CD**, SO, DS; 2014-73; TPRF 3240). One at Port Aransas, Nueces, on 22 April 2015 (**EBa**; 2015-28; TPRF 3234).

Rose-throated Becard (*Pachyramphus aglaiae*) (53). One at Santa Ana, Hidalgo, from 11–19 November 2013 (**BB**, **SG**, AMo; 2013-69; TPRF 3200). One at Bentsen SP, Hidalgo, on 28 April 2015 (**JoR**, JK; 2015-30).

Black-whiskered Vireo (*Vireo altiloquus*) (37). One at Powderhorn Ranch, Calhoun, on 1 May 2015 (**KD**, **AMu**; 2015-38; TPRF 3237).

American Dipper (*Cinclus mexicanus*) (9). One at McKittrick Canyon, Guadalupe Mountains NP, Culberson, on 29 November 2014 (**WS**, **DMu**; 2014-64; TPRF 3223).

Varied Thrush (*Ixoreus naevius*) (45). One at El Paso, El Paso, from 2 January–7 March 2015 (**JPa**; 2015-14; TPRF 3228).

Rufous-capped Warbler (*Basileuterus rufifrons*) (30). One at Chalk Bluff Park, Uvalde, from 20 April - 28 May 2013 (TF, PF, ST, CB, StC; 2013-30; TPRF 3199). One at Chalk Bluff Park, Uvalde, from 21 April–1 June 2014 (ES, StM, GPa, SeM, MV; 2014-28; TPRF 3202).

Slate-throated Redstart (*Myioborus miniatus*) (12). One at Stanford Park, Plains, Yoakum, on 19 April 2014 (**StC**; 2014-36; TPRF 3203). One at South Padre Island, Cameron, from 29–30 April 2015 (**FW**, RSt, DMc; 2015-34; TPRF 3236).

Red-legged Honeycreeper (*Cyanerpes cyaneus*) (1). One at Estero Llano Grande SP, Hidalgo, from 27–29 November 2014 (TK, ME, JO, DJ, ST; 2014-65; TPRF 3224). This represents the first documented record for Texas and the first record accepted as naturally occurring by any state's Bird Records Committee in the United States.

Golden-crowned Sparrow (*Zonotrichia atricapilla*) (38). One at South Padre Island, Cameron, on 1 May 2015 (**NL**, LA; 2015-32; TPRF 3235).

Crimson-collared Grosbeak (*Rhodothraupis celaeno*) (36). One at Frontera Audubon Thicket, Hidalgo, from 10–11 May 2014 (MG; 2014-31).

Blue Bunting (*Cyanocompsa parellina*) (46). One at Santa Ana NWR, Hidalgo, on 4 January 2015 (**DJ**; 2015-11; TPRF 3227).

Gray-crowned Rosy-Finch (*Leucosticte tephrocotis*) (1). One se. of Texline, Dallam, from 20–21 November 2014 (**DB**, **LB**, BP; 2014-59; TPRF 3221). This represents the first documented record for Texas.

Evening Grosbeak (*Coccothraustes vespertinus*) (13). Four at Wellington, Collingsworth, from 26–27 April 2014 (**KC**; 2014-53; TPRF 3219). Three at Amarillo, Potter, on 16 February 2015 (**JPr**;

2015-17; TPRF 3231). One w. of Bee Cave, Travis, on 20 March 2015 (**GLs**; 2015-23; TPRF 3232).

NOT ACCEPTED

A number of factors may contribute to a record being denied acceptance. It is quite uncommon for a record to not be accepted due to a bird being obviously misidentified. More commonly, a record is not accepted because the material submitted was incomplete, insufficient, superficial, or just too vague to properly document the reported occurrence while eliminating all other similar species. Also, written documentation or descriptions prepared entirely from memory weeks, months, or years after a sighting are seldom voted on favorably. It is important that the simple act of not accepting a particular record should by no means indicate that the TBRC or any of its members feel the record did not occur as reported. The non-acceptance of any record simply reflects the opinion of the TBRC that the documentation, as submitted, did not meet the rigorous standards appropriate for adding data to the formal historical record. The TBRC makes every effort to be as fair and objective as possible regarding each record. If the committee is unsure about any particular record, it prefers to err on the conservative side and not accept a good record rather than validate a bad one. All records, whether accepted or not, remain on file and can be re-submitted to the committee if additional substantive material is presented.

Red-necked Grebe (*Podiceps grisegena*). One near Packery Channel, Nueces, from 16 February 2013–18 February 2013 (2013-29).

Brown Booby (*Sula leucogaster*). One at Quintana Jetty, Brazoria, on 1 March 2014 (2014-18).

Arctic Tern (*Sterna paradisaea*). Two at Lake Travis/Windy Point, Travis, on 6 September 2014 (2014-49).

Buff-breasted Flycatcher (*Empidonax fulvifrons*). Four at The Bowl, Guadalupe Mountains NP, Culberson, on 14 May 2014 (2014-29).

Pacific Wren (*Troglodytes pacificus*). One at El Paso, El Paso, from 6 December 2014 - 7 December 2015 (2014-76). One at Junction, Kimble, on 1 February 2015 (2015-18).

Rufous-backed Robin (*Turdus rufopalliatus*). One at Bentsen State Park, Hidalgo, on 5 April 2015 (2015-26).

Black-vented Oriole (*Icterus wagleri*). One at Pecos, Reeves, from 22 April 2015–25 April 2015 (2015-31).

LITERATURE CITED

American Ornithologists' Union. 1998. Check-list of North American birds, 7th ed. American Ornithologists' Union, Washington, D. C.

Chesser, R. Terry, R. C. Banks, , K. J. Burns, C. Cicero, J. L. Dunn, A. W. Kratter, I. J. Lovette, A. G. Navarro-Siguenza, P. C. Rasmussen, J. V. Remsen, Jr., J. D. Rising, D. F. Stoltz, And K. Winkler, 2015. Fifty-sixth supplement to the American Ornithologists' Union Check-list of North American Birds. Auk 132:748-764.

Dittmann, D. L., And G. W. Lasley. 1992. How to document rare birds. Birding 24:145-159.

MODEL OF ENVIRONMENTAL CHARACTERISTICS OF ROADSIDE CORRIDORS AND RED- TAILED HAWK PRESENCE IN URBAN AUSTIN, TEXAS.

Jason Stayer and John T. Baccus[1,2]

Wildlife Ecology Program, Department of Biology, Texas State University, San Marcos, TX 78666

ABSTRACT.—The Red-tailed Hawk (*Buteo jamaicensis*) has shown a real ability to adapt and populate urbanized environments in some U.S. cities while displaying a negative ability to change to other environs. A population of Red-tailed Hawks in Austin, TX appears suited to roadway corridors. We assessed whether different characteristics of urbanization along roadway corridors affected occupancy by this raptor using detection and occupancy probabilities by the program PRESENCE. We modeled estimated occupancy as a function of the presence of highway medians and extent of urbanization and detection as a function of change among seasons. The relationships between median presence, vegetation, extent of urbanization and numbers of artificial perches, median presence and numbers of perches, median presence, and extent of urbanization were significantly correlated. However, the model's AIC weight did not base Red-tailed Hawk occupancy and detection solely upon these functions. Other factors not included in our model of urbanization, such as traffic flow, median dynamics, vegetative components of the median, as well as prey availability and abundance might influence occupancy and detection of Red-tailed Hawks.

The Red-tailed Hawk (*Buteo jamaicensis*) is one of the most widespread and commonly observed birds of prey in North America and provides a model for a highly adaptable, generalist predator (Preston and Beane 1993). As natural habitat declines due to the increasing effects of urbanization, a reduction in prey availability, perching, nesting, and roosting sites will cause a decrease in diversity and abundance of avian species (Brooke and Birkhead 1991). Human disturbances, habitat alteration, habitat fragmentation, and habitat loss severely affect specialist species (Jullien and Thiollay 1996). The effects of habitat loss and expansion of urbanized landscapes on the abundance of different raptor species have been mostly negative in urban areas of Colorado (Berry et al. 1998), Ohio (Dykstra et al. 2001), and Baja Mexico (Rodriguez-Estrella et al. 1998). However, conflicting correlations have resulted between raptor abundance and urbanization in different regions of the United States. Bosakowski and Smith (1997) reported that Red-tailed Hawks were more common in urban environments of New Jersey; however, Smallwood et al. (1996) detected avoidance of urban areas in Sacramento, California. For most raptor species, however, which naturally maintain large home ranges and low densities, the effects of habitat loss and alteration become apparent in urban environs (Berry et al. 1998). Most raptors display high sensitivity to urbanization (Berry et al. 1998). Raptors, however, have become more adept at using urbanized areas with ecologically important characteristics and benefit from anthropomorphic disturbances (Olendorff 1984, Bird et al. 1996). In some cases, a small proportion of hawk species might require urbanized habitat (Olendorff 1984). Swainson's Hawks (*Buteo swainsoni*) with a preference for open grassland habitat now nest in cities in Saskatchewan (James 1992). The Red-tailed Hawk appears more sensitive to the proximity of people, but displays remarkable adaptability to disturbances within urbanized landscapes (Bosakowski and Smith 1997) such as low-level

---

[1] Present address: Department of Natural Resources Management, Texas Tech University, P.O. 42125, Lubbock, Texas 79409-2125.

[2] Email: john.baccus@ttu.edu

Bull. Texas Ornith. Soc. 49(1-2): 2016

flyovers by helicopters (Andersen et al. 1989) Forman and Alexander (1998) regarded roadsides as ecological units having varied functional roles as a conduit, barrier (filter), habitat, source, and sink in the environment. Factors affecting these characters include corridor width, connectivity, usage intensity and degree of traffic flow (Forman and Alexander 1998). These corridors also create an edge effect because of fragmentation of land (Forman and Alexander 1998). There exists a paucity of research addressing roadside effects and differing degrees of urbanization on diurnal raptor abundance (Forman and Alexander 1998). Populations of Red-tailed Hawks occupy roadway corridors in Austin, Texas in urbanized environments. No study in Texas has examined the effects of the ecological features of highways and different degrees of urbanization on Red-tailed Hawk occupancy. We studied whether numbers of artificial perching sites, presence or absence of roadway medians, vegetative characteristics of medians, as well as the extent of urbanization associated with highways affected Red-tailed Hawk presence along roadways.

MATERIALS AND METHODS

We acquired presence data on Red-tailed Hawks during May 2007 to April 2008 by point counts along 6 roads in the City of Austin, Travis, Blanco and Williamson counties. The location of Austin along the Balcones Fault Escarpment of the Texas Hill Country interfaces with the Blackland Prairie. Six different roadways and 32 point-count sites occurred within our study area (Fig. 1). The lengths and numbers of point-count sites associated with highways included Mopac Expressway (39.4 km, 9), Capital of Texas Highway 360 (20.1 km, 5), Highway 620 (28.8 km, 6), Southwest Parkway (10.9 km, 2), Interregional Highway 35 (8.37 km, 2), and Hamilton Pool Road (34.4 km, 8) for a total of 142 km with a point count point at 4.44 km. We selected these roadways based on numbers of Red-tailed Hawks observed in reconnaissance of the region and a parallel distribution that would reduce the opportunity for double counting.

A rural roadway (Hamilton Pool Road) represented low-impact urbanization, and IHN 35 described high-impact urbanization. Other roadways embodied an array of different levels of urbanization and perceived raptor presence. We determined the length of each road in 2-km segments by automobile odometer. We placed a 1-km buffer in all directions at intersections with other highways. The 1-km buffer reduced the probability of double counting and ensured

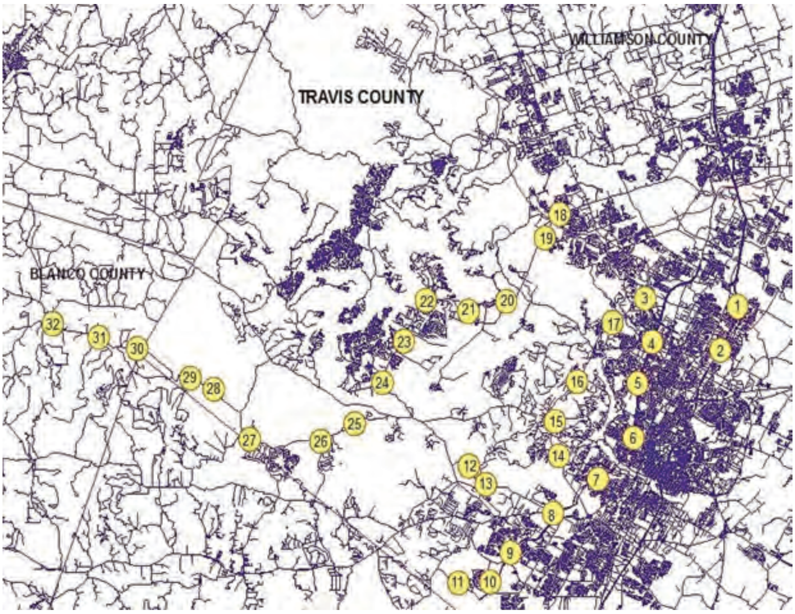


Figure 1. Map of roadways and sample sites in Travis, Blanco, and Williamson counties, Texas in 2007- 2008. The roadways and sample sites are indicated as: IHN 35 (sites 1-2), Mopac Expressway (sites 3- 11), Southwest Parkway (sites 12-13), Capital of Texas Highway 360 (sites 14-17), Highway 620 (sites 18-24), and Hamilton Pool Road (sites 25-32). Darker blue areas are considered impervious or urbanized and red lines indicate county borders.

consistent roadway and urbanization characteristics for each point.

We randomized point counts and produced a random order for sampling roadways (‘R’ 2.3.1). This random order, 8 sites/day for 4 days, followed by 2 consecutive sessions (1 month) and a new random order for the next 2 sessions. Each session consisted of counting hawks 5 min/point. We verified Red-tailed Hawk presence by observation of a hawk either perched or flying along a roadway. All sampling began 1 h post-sunrise and continued until we visited 8 sites, usually 1100 h (Skirvin 1981). We selected this period because our reconnaissance showed increased visibility in mornings due to perching by hawks.

Red-tailed Hawks perch and hunt from utility poles (Clark and Wheeler 2001). Red-tailed Hawks used artificial perch sites during morning point counts. We analyzed numbers of perching sites (utility poles, signs) located on either side of roadways and in medians as possible attractants for Red-tailed Hawk occupancy of sampling sites. We counted perch locations in both directions from each sampling site using 2 observers, a driver and passenger (Berthiaume et al. 2009). We traversed the route in 1 direction and reverse with each counting the opposite side on the second pass. The mean of the 4 surveys determined the number of perch sites for each sample site. We placed perch site totals into categories (0-50, 51-100, 101-150, 151-200, 201-250, and $>$ 250) to reduce the effect of observer bias and to reflect a diverse range of perch sites along survey routes. We analyzed whether the presence of a median affected Red-tailed Hawk occupancy by categorizing median presence as a binary variable (1 present or 0 absent regardless of size). We also observed different vegetative characteristics for roadway medians. We used simple descriptors to characterize median vegetation: (0) no vegetation, no median, (1) grass only, (2) grass and trees, (3) trees only.

We used ArcGIS 9.2 (ESRI, Redlands, CA) to determine urbanization for each point-count site. We collected data on the impervious surface from the impervious surface layer at the National Map Seamless Server (U. S. Geological Survey website [http://seamless.usgs.gov/]).

We associated impervious surfaces with roads, buildings, housing, and parking lots or an unnatural surface (http://seamless.usgs.gov/). We entered the latitude and longitude coordinates of each sample site into the impervious layer and placed a 1.0-km buffer around each location. We converted each buffer into a series of polygons to measure impervious cover and considered a value $\geq$ 13 as impervious. We derived the value $\geq$ 13 by comparing impervious surface with satellite imagery of canopy coverage (http://seamless.usgs.gov/) and ground truth of each site, while becoming familiar with point-count sites. We calculated the percent urbanization by dividing areas of polygons with a value $\geq$ 13 by the sum of areas of all polygons. We assigned terms for the extent of urbanization based on a percent of impervious cover: little ($\leq$ 10%), low (11-30%), medium-low (31-50%), medium-high (51-70%), high (71-90%), and very high (> 90%).

We used the R statistical software (Gentleman Ihaka 2006, R 2.3.1, Free Software Foundation, Inc., Boston, MA) to assess differences in artificial perch counts between observers by paired *t*-tests. We used Pearson’s correlation coefficient to determine the strength of relationships among urbanization characteristics. We used Program PRESENCE (Ver. 2.4, USGS) to analyze point count data to determine occupancy of Red-tailed Hawks at each sample site. We used occupancy models to examine each sample site as a function of median presence and the extent of urbanization. We did not analyze characteristics indicating the degree of urbanization with a strong positive relationship ($r > 0.60$) because similar models resulted via the Akaike Information Criteria (AIC). We assessed characteristics indicating the extent of urbanization with weak relationships ($r \leq$ 60) using the AIC to determine whether Red-tailed Hawk occupancy along roadways was affected by different characteristics of urbanization.

We analyzed single and multiple season models to assess whether differences in occupancy probabilities (ψ) existed throughout a season or fluctuated seasonally. During the assessment of multiple-season models, we maintained colonization (γ) and extinction (ε) probabilities constant (.) to eliminate the effects immigration and emigration might have on Red-tailed Hawk occupancy. Detection probabilities (p) for single- and multiple-season models were either held constant (.), assuming detection for all sites was equivalent throughout the season or detection probability was a function of seasonality (seasonal). We selected models based upon their AIC weight in comparison to other models. Results were considered significant at $P = 0.05$.

RESULTS

We had 60 detections of Red-tailed Hawks in 768 point counts (7.8%); we recorded 58 detections as a perch and only 2 as flying. Artificial substrates formed the site for 56 perch events and only 2 perches on natural substrates. Insignificant observer bias occurred in perch counts ($t = 1.554$, P = 0.065, Table 1); however, drivers had difficulty giving complete attention to counting perches while maintaining focus on traffic conditions. The most artificial perches for a site was at site 1 ($\bar{X} = 250.5$, SE = 5.5). Site 12 had no perches (Table 1). The IHN-35 roadway contained the highest number of perches ($\bar{X} = 179$, SE = 71.5) and Southwest Parkway provided the fewest perches ($\bar{X} = 15$, SE = 15.0).

The relationships between median presence and vegetation ($r = 0.721$, $P > 0.001$), extent of urbanization and numbers of artificial perches ($r = 0.746$, $P > 0.001$), median presence and numbers of perches ($r = 0.405$, $P = 0.01$), and median presence and extent of urbanization ($r = 0.391$, $P = 0.013$) were significantly correlated. The relationships between extent of urbanization and vegetation ($r = 0.189$, $P = 0.151$) and numbers of artificial perches and vegetation ($r = 0.136$, $P =$

Table 1. Total artificial perching sites on roadway corridors in Austin, Texas assessed by two observers (Obs A, B) per sample site, and mean number of artificial perches at each site.

Sample Site	Roadway	Obs A	Obs B	Mean
1	IHN-35	256	245	250.5
2	IHN-35	110	105	107.5
3	MOPAC	146	130	138
4	MOPAC	141	153	147
5	MOPAC	204	188	196
6	MOPAC	178	178	178
7	MOPAC	166	137	151.5
8	MOPAC	156	126	141
9	MOPAC	66	64	65
10	MOPAC	5	23	14
11	MOPAC	7	27	17
12	SW PKWY	0	0	0
13	SW PKWY	32	28	30
14	COT 360	96	92	94
15	COT 360	32	37	34.5
16	COT 360	56	51	53.5
17	COT 360	82	52	67
18	HGWY 620	105	109	107
19	HGWY 620	89	78	83.5
20	HGWY 620	65	61	63
21	HGWY 620	51	55	53
22	HGWY 620	72	80	76
23	HGWY 620	81	86	83.5
24	HGWY 620	75	82	78.5
25	HAM POOL	35	28	31.5
26	HAM POOL	16	16	16
27	HAM POOL	19	19	19
28	HAM POOL	27	26	26.5
29	HAM POOL	11	10	10.5
30	HAM POOL	20	13	16.5
31	HAM POOL	24	23	23.5
32	HAM POOL	17	16	16.5

0.23) were not significantly correlated. Based on the high significance of correlations between median presence and vegetation and extent of urbanization and numbers of artificial perches, we analyzed these characteristics by AIC. Median presence was the primary characteristic determining presence of vegetation. Medians provided a substantial amount of area for growth of vegetation. The extent of urbanization was the primary reason for more artificial perches since more utility poles would be necessary to service residents (Table 2).

The model containing occupancy as a function of median presence and detection probability as a function of seasonality resulted in the best fit among 15 models (Table 3). The single season model that assessed occupancy as a function of both median presence and extent of urbanization and detection probability as a function of seasonality produced unreliable data and disregarded. Sites 1-17 occurred on roadways with presence of medians, and sites 18-32 were on roadways with no median. Figure 2 displays 95% confidence intervals for estimates of occupancy as a function of median presence (ψ = 0.96, SE = 0.09) and median absence (ψ = 0.58, SE = 0.141). Figure 3 displays 95% confidence intervals for detection probabilities for season 1 (May-Jul, P = 0.8, SE = 0.2), season 2 (Aug-Oct, P = 0.16, SE = 0.03), season 3 (Nov- Jan, P = 0.11, SE = 0.03), and season 4 (Feb-Apr, P = 0.05, SE = 0.02).

Table 2. Median presence (Y = yes, N = no), degree of urbanization (%), and vegetation type found on median (Grass, Trees, Mix-grass and trees, NA-not applicable) at sample sites on roadways in Travis, Blanco, and Williamson counties.

Sample Site	Median Presence	Degree of Urbanization	Median Vegetation
1	Y	74.47%	GRASS
2	Y	38.46%	GRASS
3	Y	70.22%	GRASS
4	Y	79.70%	GRASS
5	Y	54.31%	MIX
6	Y	39.44%	GRASS
7	Y	45.34%	GRASS
8	Y	54.29%	GRASS
9	Y	54.09%	MIX
10	Y	24.72%	MIX
11	Y	9.78%	TREES
12	Y	5.01%	TREES
13	Y	11.18%	GRASS
14	Y	35.13%	GRASS
15	Y	13.82%	GRASS
16	Y	39.26%	GRASS
17	Y	19.76%	GRASS
18	N	72.63%	NA
19	N	35.03%	NA
20	N	5.69%	NA
21	N	17.46%	NA
22	N	24.12%	NA
23	N	21.30%	NA
24	N	27.39%	NA
25	N	27.25%	NA
26	N	0.22%	NA
27	N	4.16%	NA
28	N	0.33%	NA
29	N	0.37%	NA
30	N	0.73%	NA
31	N	0.20%	NA
32	N	0.31%	NA

DISCUSSION

Model analyses indicated that occupancy probability as a function of median presence and detection probability as a function of seasonality were the best estimators for the presence of Red-tailed Hawks in Austin, TX. Occupancy expectations

Table 3. Akaike Information Criteria (AIC), AIC weights (w) and number of parameters (K) of PRESENCE models analyzing Red-tailed Hawk occupancy (ψ) and detection probability (p) along roadway corridors in Austin, Texas as functions of median presence (median), degree of urbanization (urban), both (median-urban), or a constant (.). Colonization (γ) and extinction (ε) probabilities held constant in multiple season models.

Model	AIC	w	K
ψ(median), p(seasonal)*	410.95	0.41	6
ψ(.), p(seasonal)	413.93	0.09	5
ψ(.), γ(.), ε(.), p(.)	414.06	0.09	4
ψ(.), γ(.), ε(.), p(seasonal)	414.85	0.07	7
ψ(urban), p(seasonal)	414.85	0.06	6
ψ(median), p(.)	415.20	0.05	3
ψ(median-urban), p(.)	415.29	0.05	3
ψ(urban), γ(.), ε(.), p(.)	415.36	0.05	5
ψ(median), γ(.), ε(.), p(.)	415.72	0.04	5
ψ(urban), γ(.), ε(.), p(seasonal)	415.86	0.04	8
ψ(median), γ(.), ε(.), p(seasonal)	416.27	0.03	8
ψ(median-urban), γ(.), ε(.), p(.)	416.90	0.02	5
ψ(median-urban), γ(.), ε(.), p(seasonal)	417.07	0.02	8

*Model selected.

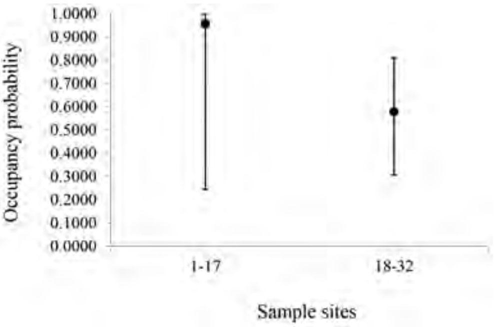


Figure 2. Occupancy probability of Red-tailed Hawks as a function of median presence and their 95% confidence intervals along roadway corridors in Austin, Texas.

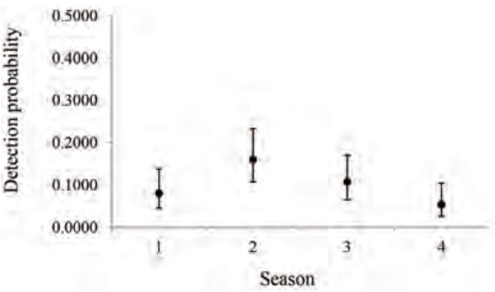


Figure 3. Detection probability of Red-tailed Hawks as a function of seasonality and their 95% confidence intervals along roadway corridors in Austin, Texas. Season 1 (May 07–Jul 07), season 2 (Aug 07–Oct 07), season 3 (Nov 07–Jan 08), and season 4 (Feb 08–Apr 08).

were higher with median presence than absence, but displayed large confidence intervals in comparison (Fig. 1). Detection probabilities varied seasonally with the highest detection in Aug-Oct (Fig. 2) during the peak of fall migration for Red-tailed Hawks (Allen 1993) and could explain the higher detection probability. Although this model had the best fit for the data, the AIC weight (w = 0.41) leaves room for questions concerning occupancy.

Encroachment of urbanization into natural habitats has the greatest effects on specialist raptor species (Jullien and Thiollay 1996) due to habitat loss, alteration, and human disturbance. However, generalist species, such as the Red-tailed Hawk, benefit from this encroachment in areas retaining ecologically important features (Bird et al. 1996). It is unknown whether prey abundance is affecting occupancy of this raptor in Austin due to the ability of the Red-tailed Hawk to incorporate diet-switching behaviors (Steenhof and Kochert 1988). Olendorff (1984) suggested some raptor species might require a more urbanized habitat in some geographic regions of the United States. This use of urban areas may result from the extent of occupancy of the optimum natural habitat (source) and roadside habitats retaining ecologically important characteristics may act as sinks. These urban areas appear to benefit Red-tailed Hawks, a species generalist, in using a less favorable habitat.

It seems one cannot solely assess occupancy probability of Red-tailed Hawks in Austin by characteristics examined in our study. However, occupancy as a function of median presence is a good place to start when considering different factors leading to the presence of Red-tailed Hawks. Andersen et al. (1989) viewed the ability of Red-tailed Hawks to adapt to intense helicopter traffic, and thus, should be able to adapt to areas of high traffic and noise. This characteristic of roadside ecology needs more examination (Forman and Alexander 1998) and might have a larger role in occupancy. Forman and Alexander (1998) also indicated that the fragmentation effects of urbanization and roadway ecosystems on occupancy probability need further study.

Watson and Simpson (2014) found raptor abundance on rural roads had a strong negative correlation with increasing traffic levels in Hays, Caldwell, Gonzales, and Guadalupe counties. The abundance of Red-tailed Hawks observed on rural roadways was almost 3-fold larger than the number of Red-tailed Hawks detected on urban roadways in Austin.

A larger sample size would allow more focus on vegetative components, such as type, dynamic structure, and edge effect on and immediately surrounding roadways. Urban encroachment will affect natural nesting sites for all raptor species and should be examined in future research as well as the abundance of artificial nesting sites to determine the effects on hawk occupancy of roadway corridors. Factors such as median dynamics and prey abundance are research areas that might lead to stronger conclusions concerning occupancy and detection probabilities of Red-tailed Hawks.

The results our research indicated that Red-tailed Hawks in Austin are probably adapting to roadside corridors due to a variety of factors. This geographic locale likely biased these factors estimating occupancy. With further research into geographically similar areas, one can assess the factor(s) that contribute to Red-tailed Hawk occupancy along roadways. This information can assist in the construction of new highways or scenic byways advancing into natural areas (i.e., Balcones Canyonlands Reserve) to retain ecologically important features of hawk habitat. If top carnivores are beginning to fill niches created by increasing urban sprawl, then we should use this opportunity to examine the reasons why this is occurring and adjust methods of road construction into the natural habitat of raptors to allow for the highest ecological diversity possible.

LITERATURE CITED

Allen, P. E. 1993. Raptor migration count trends at Hawk Mountain (1934–1991). Unpublished Manuscript, Hawk Mountain Sanctuary, Kempton, Pennsylvania.

Andersen, D. E., O. J. Rongstad, and W. R. Mytton. 1989. Response of nesting Red-Tailed Hawks to helicopter overflights. Condor 91: 296-299.

Berry, M. E., C. E. Bock, and S. L. Haire. 1998. Abundance of diurnal raptors on open space grasslands in an urbanized landscape. Condor 100: 601-608.

Berthiaume É., M. Bélisle, and J. Savard. 2009. Incorporating detectability into analyses of population trends based on hawk counts: a double-observer approach. Condor 111: 43– 58.

Bird, D. M., D. E. Varland, and J. J. Negro. 1996. Raptors in human landscapes: adaptations to built and cultivated environments. Academic Press, San Diego, California.

Bosakowski, R. and D. G. Smith. 1997. Distribution and species richness of a forest raptor community in relation to urbanization. Journal Raptor Research 31: 26-33.

Brooke, M. and T. Birkhead. 1991. Factors indicating and influencing bird distributions. Pp. 170-171 in The Cambridge Encyclopedia of Ornithology (M. Brooke and T. Birkhead Editors.) Cambridge University Press, Cambridge, Massachusettes.

Clark, W. S. and B. K. Wheeler. 2001. Red-tailed Hawk. Pages. 213-222 in Peterson field guides: hawks of North America (W.S. Clark and B.K. Wheeler, Editors). Second Edition. Houghton Mifflin Co., New York.

Dykstra, C. R., F. B. Daniel, J. L. Hays, and M. M. Simon. 2001. Correlation of Red-shouldered Hawk abundance and macrohabitat characteristics in southern Ohio. Condor 103: 652-656.

Forman, R. T. T. and L. E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29: 207-231.

Gentleman, R. and R. Ihaka. 2006. R Project for Statistical Computing. Version 2.3.1 Statistics Department of the University of Auckland, Auckland, New Zealand. http://www.r-project.org.

James, P. C. 1992. Urban-nesting of Swainson's Hawks in Saskatchewan. Condor. 94: 773-774.

Jullien. M. and J. M. Thiollay. 1996. Effects of rain forest disturbance and fragmentation: comparative changes of the raptor community along natural and human-made gradients in French Guiana. Journal of Biogeography 23: 7-25.

Olendorff, R. R. 1984. Land management for raptor conservation: 1984-2034. Pages 49-52 in Raptor conservation in the next 50 years (S. E.Senner, C.M. White, and J.R. Parrish, Editors). Raptor Research Foundation, Raptor Research..

Preston, C. R. and R. D. Beane. 1993. Red-tailed Hawk (*Buteo jamaicensis*), The Birds of North America. Number 52.

Rodriguez-Estrella, R., J. A. Donazar, and F. Hiraldo. 1998. Raptors as indicators of environmental change in the scrub habitat of Baja California Sur, Mexico. Conservation Biology 12:921-925.

Skirvin, A. A. 1981. Effect of time of day and time of season on the number of observations and density estimates of breeding birds. Studies in Avian Biology 6:271-279.

Smallwood, S. K., B. J. Nakamoto, And S. Geng. 1996. Association analysis of raptors on a farming landscape: Pages 177-190, in Raptors in human landscapes: adaptations to built and cultivated environments. (D. M. Bird, D. E. Varland, and J. J. Negro, Editors). Academic Press, San Diego, California.

Steenhof, K. and M. N. Kochert. 1988. Dietary responses of three raptor species to changing prey densities in a natural environment. Journal of Animal Ecology 57:37-48.

United States Geological Survey, The National Map Seamless Server; available from http://seamless.usgs.gov/; (accessed 19 March 2008).

Watson, K. A. and T. R. Simpson. 2014. Relationship of vehicular traffic flow and roadside raptor and vulture abundance in south-central Texas. Bulletin of the Texas Ornithological Society 47:17-23.

MULTIPLE PATERNITY AND OFFSPRING SEX RATIO IN REDDISH EGRETS (*EGRETTA RUFESCENS*)

Austin Hill and M. Clay Green[1]

Department of Biology, Population and Conservation Program,
Texas State University-San Marcos, San Marcos, Texas 78666, USA

ABSTRACT.—Using microsatellites, we assessed the occurrence of multiple paternities in Reddish Egret (*Egretta rufescens*) broods and determined the sex ratio among nest clutches and across the study population. We collected genetic samples from 326 individual nestlings in 8 breeding colonies in the Laguna Madre of Texas and Tamaulipas, Mexico; all 326 individuals were used for population estimate of nestling sex ratio. In 52 nests across 8 breeding colonies, we collected blood samples from the entire clutch to assess paternity and sex ratios within each nest. We detected evidence of multiple paternity in Reddish Egret clutches, occurring in 29.6% of nests with > 3 eggs and in 15.4% of all clutch sizes. Sex ratio among nests where all individuals from the nest were sampled and from all individuals in this study was 58 males to 80 females and 146 males to 180 females respectively. The frequency of males across the population (44.8%) and within entire clutches (42.0%) was not significantly different from the expected 1:1 sex ratio (Population: $\chi^2 = 3.37$, $P = 0.066$; Within clutches: $\chi^2 = 1.32$; $P = 0.251$). Despite presumed monogamy, multiple paternities from extra pair copulations (EPC) do occur in Reddish Egrets and are most prevalent in nests with all white morph nestlings, the less common morph in Texas. The offspring sex ratio did not significantly deviate from the expected 1:1 ratio suggesting that none of the potential mechanisms for sex ratio adjustment are occurring within Reddish Egrets.

Multiple paternity has been observed in numerous taxa (Avise 1994) though for many years it was thought that most bird species were monogamous (Lack 1968). Recent advances in molecular techniques have revealed that social monogamy may not correspond to genetic monogamy. Many colonial nesting birds including those in the order Ciconiiformes have exhibited extra-pair copulation and multiple paternity (Gladstone 1979, McKilligan 1990, Frederick 1987, Ramo 1993, Krebs et al. 2004, Wei et al. 2005, Miño et al. 2009).

Many adult bird species exhibit no significant sexual dimorphism between males and females. Sexing individuals with molecular techniques can provide a quick and accurate method for determining an individual's sex (Griffiths et al. 1998, Lessels and Mateman 1998, Fridolfsson and Ellegren 1999). An offspring sex ratio different from 1:1 can be suggestive of the female condition in relation to habitat quality or male fitness (Svennson and Nilsson 1996, Nager et al. 1999, Kolliker et al. 1999).

Reddish Egrets (*Egretta rufescens*) have an estimated global population of 5,000-7,000 (Paul 1991, Green 2006, Wilson et al. 2014). They are the rarest heron in North America, with populations still recovering from the plume trade of the early 20th century that nearly extirpated the species from Gulf Coast states; these egrets are considered "threatened" and a "species of special concern" in Texas and Florida, respectively (Lowther and Paul 2002). Several gaps in the knowledge of the ecology of Reddish Egrets exist in part due to its rarity. Information on multiple paternity and offspring sex ratio that could contribute to the conservation and management in this species is lacking. Reddish Egrets are thought to be monogamous, but several other species in the family Ardeidae have shown to engage in extra pair copulations (Ramo 1993, Krebs et al. 2004, Wei et al. 2005). Additionally, the study

---

[1] Email: claygreen@txstate.edu

of multiple paternity in the plumage dimorphic Reddish Egret may shed some light on the phenomenon of color polymorphism if differences in multiple paternity are detected between white and dark morph nests (Green 2005; Holderby et al. 2012; Holderby and Green 2013). The objectives of our study were to examine multiple paternity in reddish egret nest-mates as well as determine the offspring sex ratios.

METHODS

We collected blood samples from 326 nestlings in eight colonies from April–July 2006, March–July 2007, and March–July 2008. Sampling locations spanned the Texas/Mexico Gulf coast along the Laguna Madre of Texas and the state of Tamaulipas, Mexico as well as breeding colonies in Great Inagua, Bahamas and Baja California Sur, Mexico. Blood was obtained from the brachial vein of nestlings using a 25-gauge needle to puncture the vein. We collected ~4 µl of blood from each bird using a capillary tube and then placed the blood in a vial containing 600 µl of cell lysis solution (Puregene, Gentra Systems). For 52 nests (n = 138 individuals), we collected blood samples from the entire brood to assess paternity and sex ratios; the remaining blood samples came from individuals (n = 188) as part of a larger study on gene flow across the range of the species (Hill et al. 2012). The complete brood (nest) samples came from 52 nests from Baja California (n = 3), Florida (n = 4), Bahamas (n = 6) and Texas/Tamaulipas (n = 39). We defined the color morph of each nest based on the observation of nestling plumage color. Nests defined as white or dark morph nests contained only white or dark morph nestlings, respectively. Mixed-morph nests had nestlings of both color morphs.

We performed DNA extraction using Puregene (Qiagen) DNA isolation protocol for avian whole blood. We screened 13 microsatellite loci primers and found 12 that could be successfully amplified in all 326 samples (Hill and Green 2011, Hill et al. 2012). Due to the presence of null alleles in three loci, we used the remaining nine loci for all further analyses (Hill and Green 2012). Individuals were sexed using primers 2250F and 2718R as proposed by Fridolfsson and Ellegren (1999). PCR reactions followed the methods of Fridolfsson and Ellegren (1999), but quantities were adjusted to 50 µl to accommodate the BIO-RAD PTC-100 thermocycler. The thermal sequence of the thermocycler was modified from the Fridolfsson and Ellegren (1999) method by removing the touchdown sequence with the protocol as follows: 94°C for 2 min, 35 cycles of 94°C for 30 s, 57°C for 30 s, 72°C for 30 s, and a final extension at 72°C for 5 min. Products were then run on a 2% agarose gel for manual scoring. As females in birds are the heterogametic sex, females will show two bands while males will have only one band.

GERUD 2.0 (Jones 2005) was used to assess the minimum number of sires in a complete clutch when maternal genotype is unknown by reconstructing the maternal genotype from the progeny genotypes. Offspring sex ratios were calculated for individuals within complete clutches and across all sampling sites. The Chi-square Goodness of fit test was used to compare offspring sex ratio with the expected 1:1 sex ratio.

RESULTS

Output from GERUD 2.0 (Jones 2005) suggested that 8 of 52 (15.4%) broods had a minimum of two males contributing to offspring genotypes. Brood size from nests where blood was taken from all nestlings varied from 2 to 5 individuals. Among color morphs, percentages of nests with multiple paternity were 8.3% (2 of 24 nests) for dark morph, 30.8% (4 of 13 nests) for white morph, and 13.3% (2 of 15 nests) for mixed morph. Because power to detect multiple paternity in clutch/brood sizes of ≤ 2 is poor, we also assessed mixed paternity in broods ≥ 3 for further analysis. Among complete nests that had ≥ 3 nestlings, 8 of 27 (29.6%) nests showed evidence for multiple paternity; no broods of ≤ 2 nestlings showed evidence for multiple paternity. Multiple paternity across color morphs in ≥ 3 nestling broods revealed 28.6 % (2 of 7 nests), 57.2% (4 of 7 nests) and 15.4% (2 of 13 nests) for dark, white and mixed morph nests respectively.

The sex ratio for nests where all nestlings were sampled ($N = 52$ nests) was 58 males to 80 females. Sex ratio among all 326 nestlings sampled was 146 males and 180 females. The frequency of males across the population (44.8%) and within entire clutches (42.0%) was not significantly different from the expected 1:1 sex ratio (Population: $\chi^2 = 3.37$, $P = 0.066$; Within clutches: $\chi^2 = 1.32$; $P = 0.251$).

DISCUSSION

Despite presumed monogamy (Lowther and Paul 2002), multiple paternity occurred in ~15% of all Reddish Egret nests examined. Our estimate of the frequency of multiple paternity in Reddish Egrets is probably conservative. We did not detect multiple paternity in broods of ≤ 2 nestlings, presumably because it is not difficult to reconstruct only one maternal and one paternal genotype to match offspring genotypes. Even if multiple loci had four different alleles, the combination could still possibly come from only two adults. Thus, the actual percentage of nests with multiple paternity may be closer to the estimated ~30% of nests with three nestlings. In a paternity study with Roseate Spoonbills (*Platalea ajaja*), another colonial nesting waterbird, the occurrence of half siblings in clutches indicated multiple sires were present in 12 of 28 (42.8%) nests from a natural population (Miño et al. 2009). In Chinese Egrets (*Egretta eulophotes*) attempted EPCs were successful 77% of the time (Wei et al. 2005). Extra-pair copulations may be more common in Reddish Egret than originally recognized, given evidence in other Ardeid species' (Gladstone 1979, Wei et al. 2005).

Among color morph nests, nests with all white nestlings exhibited the highest rate of multiple paternity; 5 of the 8 (62.5%) clutches documented with half siblings were white morph nests. Evidence for multiple paternity was less prevalent in dark and mixed morph nests; the only mixed-morph nest exhibiting evidence of half siblings was a 5 egg clutch (4 dark morph and 1 white morph nestling). Majority (6 of 8) nests with multiple paternity came from Texas samples. White morphs are the rarer morph along the Texas/Mexico coast based on estimates of ~35% white morphs documented nesting in colonies (A. Hanna, unpubl. data). It is unclear why the rarer morph might have higher rates of multiple paternity but could be due to higher density of nesting Reddish Egrets (regardless of morphs present) or for other reasons yet unexplained. While our samples sizes for Florida and the Bahamas were low (n = 10 nests), we only detected multiple paternity in one nest (Great Inagua, Bahamas) and densities of nesting Reddish Egrets are much lower in Florida and Bahamas (Green et al. 2011). Future studies should evaluate the prevalence of multiple paternity across the geographic range of the species to assess if multiple paternity varies based on nesting density, color morph ratio, or other ecological and environmental factors.

The offspring sex ratio did not significantly deviate from the expected 1:1 ratio suggesting that none of the potential mechanisms for sex ratio adjustment are occurring within Reddish Egrets (Krackow 1995). While sex ratio at birth could vary due to female condition or environmental conditions, we found no evidence for any deviation occurring in the populations we sampled. However, future studies on sex ratios at hatch and potential differential survival of males and females during brooding and post-fledging stage are warranted as there is some indication of sex-biased dispersal and survival in juvenile Reddish Egrets (Geary et al. 2015).

ACKNOWLEDGMENTS

We thank Chris Nice, Jim Ott, David Newstead, LeRoy Overstreet, Hector Quintinilla, Alfonso Banda, Mark Biel, James Hall, Zachary Holderby, William Simper, Jeff Troy, Bart Ballard, Elizabeth Bates, Eduardo Palacios, Lynn Gape, Henry Nixon, Predensa Moore, Coastal Bend Bays and Estuaries Program, Audubon Texas, Texas General Land Office, ProNatura, Bahamas National Trust and Padre Island National Seashore. This manuscript is part of larger study on nesting and foraging ecology of Reddish Egrets funded by U.S. Fish and Wildlife Service Regions 2 and 4.

LITERATURE CITED

Avise, J. C. 1994. Molecular markers, natural history and evolution. Chapman and Hall Inc., London, United Kingdom.

Frederick, P. C. 1987. The response of male White Ibises to their mate's extra-pair copulation. Behavioral Ecology and Sociobiology 21:223-228.

Fridolfsson, A. K., and H. Ellegren. 1999. A simple and universal method for molecular sexing of non-ratite birds. Journal of Avian Biology 30:116-121.

Gladstone, D. E. 1979. Promiscuity in monogamous colonial birds. American Naturalist 114:545-557.

Geary, B., B. Ballard and M. C. Green. 2015. Movements and survival of juvenile reddish egrets on the Gulf Coast. Endangered Species Research 28: 123-133.

Green, M. C. 2005. Plumage dimorphism in the reddish egret: does plumage coloration influence foraging habitat use and tactics? Waterbirds 28(4): 519-524.

GREEN, M. C. 2006. Status report and survey recommendations on the Reddish Egret (*Egretta rufescens*). Pages 1-37 U.S. Fish and Wildlife Service, Atlanta, Georgia.

GREEN, M. C., A. HILL, Z. HOLDERBY, J. TROY AND B. GEARY. 2011. Status of breeding Reddish Egrets on Great Inagua, Bahamas with comments on breeding territoriality and the effects of hurricanes. Waterbirds: 34: 213 – 217.

GRIFFITHS, R., M. C. DOUBLE, K. ORR, AND R. J. G. DAWSON. 1998. A DNA test to sex most birds. Molecular Ecology 7:1071-1075.

HILL, A., AND M. C. GREEN. 2011. Characterization of 12 polymorphic microsatellites for the Reddish Egret, *Egretta rufescens*. Conservation Genetic Resources 3:13-15.

HILL, A., E. PALACIOS AND M. C. GREEN. 2012. Genetic diversity and population structure of North America's rarest heron, the Reddish Egret (*Egretta rufescens*). Conservation Genetics 13:535-543.

HOLDERBY, Z., W. SIMPER, B. GEARY, AND M. C. GREEN. 2012. Potential factors affecting nest initiation date, clutch size, and nest success in the plumage dimorphic Reddish Egret. Waterbirds 35: 437-442.

HOLDERBY, Z. AND M. C. GREEN. 2013. Plumage dimorphism and nest site selection of Reddish Egrets (*Egretta rufescens*) in the Laguna Madre, Texas. Bulletin of the Texas Ornithological Society 46: 1-9.

JONES, A. G. 2005. GERUD 2.0: a computer program for the reconstruction of parental genotypes from half-sib progeny arrays with known or unknown parents. Molecular Ecology Notes 5:708-711.

KÖLLIKER, M., P. HEEB, I. WERNER, A. C. MATEMAN, C. M. LESSELS, AND H. RICHNER. 1999. Offspring sex ratio is related to male body size in the Great Tit (*Parus major*). Behavioral Ecology 10:68-72.

KRACKOW, S. 1995. Potential mechanisms for sex ratio adjustment in mammals and birds. Biological Reviews 70:225-241.

KREBS, E. A., W. HUNTE, AND D. J. GREEN. 2004. Plume variation, breeding performance and extra-pair copulations in the Cattle Egret. Behaviour 141:479-499.

LESSELS, C. M., AND A. C. MATEMAN. 1998. Sexing birds using randomly amplified polymorphic DNA (RAPD) markers. Molecular Ecology 7:187-195.

LOWTHER, P. E., AND R. T. PAUL. 2002. Reddish Egret (*Egretta rufescens*). The Birds of North America, Online: http://bna.birds.ornell.edu/bna/species/633. (accessed October 27, 2016).

MCKILLIGAN, N. G. 1990. Promiscuity in the Cattle Egret (*Bubulcus ibis*). Auk 107:334-341.

MIÑO, C. I., G. M. SAWYER, R. C. BENJAMIN, AND S. N. DEL LAMA. 2009. Parentage and relatedness in captive and natural populations of the Roseate Spoonbill (*Aves: Ciconiiformes*) based on microsatellite data. Journal of Experimental Zoology 311A:453–464.

NAGER, R. G., P. MONOGHAN, R. GRIFFITHS, D. C. HOUSTON, AND R. DAWSON. 1999. Experimental demonstration that offspring sex ratio varies with maternal condition. Proceedings of the National Academy of Sciences 96:570-573.

PAUL, R. T. 1991. Status report—*Egretta rufescens* (Gmelin) Reddish Egret. U.S. Fish and Wildlife Service, Houston, Texas.

RAMO, C. 1993. Extra-pair copulations of Grey Herons nesting at high densities. Ardea 81:115–120.

SVENNSON, E., AND J. Å. NILSSON. 1996. Mate quality affects offspring sex ratio in Blue Tits. Proceedings of the Royal Society B 263:357-361.

WEI, G. A., Z. H. YIN, AND F. M. LEI. 2005. Copulations and mate guarding of the Chinese Egret. Waterbirds 28:527–530.

WILSON, T. E., J. WHEELER, M. C. GREEN AND E. PALACIOS. 2014. Reddish Egret Conservation Action Plan. Reddish Egret Conservation Planning Workshop, October 2012. Corpus Christi, Texas.

AN OBSERVATIONAL STUDY OF CARRION USE BY FORAGING TURKEY VULTURES (*CATHARTES AURA*) IN WEST TEXAS

Steven G. Platt[1,5], Thomas R. Rainwater[2,4], and Stanlee M. Miller[3]

[1]*Department of Biology, Box C-64, Sul Ross State University, Alpine, TX 79832*
[2]*Tom Yawkey Wildlife Center & Baruch Institute of Coastal Ecology and Forest Science, Clemson University, P.O. Box 596, Georgetown, SC 29442*
[3]*Department of Biological Sciences, 132 Long Hall, Clemson University, Clemson, SC 29634-1903*

ABSTRACT.—We investigated carrion use and foraging ecology of Turkey Vultures (*Cathartes aura*) in the Trans-Pecos of West Texas from 2006-2011. Mammals (particularly ungulates) were the primary source of carrion for vultures. Most carrion consumed by vultures was provided by wildlife-vehicle collisions. Turkey vultures were more likely to be found at fresh and bloated carcasses than those in advanced stages of decay, and there was a significant positive correlation between carcass mass and group size of feeding vultures. Vultures use different processing strategies on large and small carcasses. The absence of small carrion in the diet and large feeding aggregations of Turkey Vultures may be due in part to the rarity of Black Vultures (*Coragyps atratus*) in our study area. Novel food items found in this study include deer antlers and porcupine carrion. Feeding trials suggest visual cues may be more important for detecting carrion than previously recognized. The discovery of a Turkey Vulture killed while feeding on a gray fox carcass represents one of the few instances of mortality attributable to predation.

The Turkey Vulture (*Cathartes aura*) is the most widely distributed vulture in the New World, ranging from southern Canada to southernmost South America (Kirk and Mossman 1998). Turkey Vultures are habitat generalists and soaring, obligate scavengers that opportunistically feed on a wide range of wild and domestic carrion (Kirk and Mossman 1998; Ruxton and Houston 2004). As scavengers, Turkey Vultures accelerate carrion decomposition rates and transfer energy across trophic levels, ultimately increasing the stability of ecological communities and food webs (DeVault et al. 2003; Barton et al. 2013; Allen et al. 2014). Turkey Vultures and other avian scavengers also provide critical, but underappreciated ecosystem services by removing carrion, regulating diseases, and cycling nutrients (Wenny et al. 2011). Although the diet of Turkey Vultures has in general, been well-documented (Kirk and Mossman 1998 and references therein; Rhodes 2007; Platt and Rainwater 2008; Platt et al. 2014; Olson et al. 2016), patterns of food consumption appear to vary across the geographic range, among habitats, and by season (Rhodes 2007). Furthermore, few dietary studies have been conducted in regions where Turkey Vultures are allopatric with respect to other vultures (Prior 1990; Estrella 1994).

Herein we characterize the diet, report the consumption of novel food items, describe carcass processing, and comment on other aspects of the trophic ecology of Turkey Vultures in West Texas, USA. Information on diet and feeding behavior is fundamental to understanding the ecology of any organism (Rosenberg and Cooper 1990). In addition, an improved understanding of Turkey Vulture biology, including their prandial habits is of interest to conservation agencies tasked with managing and protecting vulture populations (DeVault et al. 2005; Rhodes 2007).

STUDY AREA

Our study area was located within the Trans-Pecos ecological region of Texas and encompassed parts of Brewster, Presidio, and Jeff Davis counties; however,

---

[4] Email: Corresponding author: trrainwater@gmail.com

[5] Current address: Wildlife Conservation Society – Myanmar Program, Building C-1, Aye Yeik Mon 1st Street, Hlaing Township, Yangon, Myanmar

most observations were made in the vicinity (≤ 100 km) of Alpine (30° 21' N; 103° 39' W) in Brewster County. The Trans-Pecos is characterized by mild winters and hot summers, with most precipitation (mean annual rainfall ca. 370 mm) occurring during a relatively brief monsoonal period extending from June through August (Leithead 1959). Elevations range from ca. 300 m along the Rio Grande to 2555 m in the Davis Mountains (Powell 1998). Major habitat types correspond to changes in elevation: desert scrub and arid grassland (*Bouteloua gracilis*, *Prosopis glandulosa*, and *Larrea tridentata*) occur at lower elevations, with oak-juniper-pinyon woodland (*Quercus* spp., *Juniperus* spp., and *Pinus cembroides*) at mid-elevations, and coniferous forest (*Pinus* spp., *Pseudotsuga menziesii*) at the highest elevations (Powell 1998). Cattle ranching is the predominant use of private land (Powell 1998), and chronic overgrazing has resulted in the conversion of grassland to shrub desert in many areas (Leithead 1959; Wilde and Platt 2011). Turkey Vultures generally arrive in our study area during late February or early March (ingress from East) and depart in mid-October (Platt and Rainwater, unpubl. data). Black Vultures (*Coragyps atratus*) are rare throughout the Trans-Pecos Region (Petersen and Zimmer 1998), and in our study area were only encountered in small numbers along the Rio Grande (SGP, pers. obs.). To avoid redundancy in the text, "vulture" refers to Turkey Vulture unless otherwise stated.

METHODS

Most observations of foraging Turkey Vultures were made when carcasses were opportunistically discovered as we engaged in other fieldwork or routine activities (e.g., driving and bicycling) from September 2006 through April 2011. At each carcass, we assessed the stage of decomposition (fresh, bloated, active decay, advanced decay, and dry) according to Payne (1965), counted the number of vultures present, and recorded evidence of vulture feeding activity. If the carcass was that of a road-killed animal, we noted injuries attributable to vehicle collision. When possible, we made follow-up visits to carcasses at intervals ≥ 24 hrs. During these visits, we reassessed the stage of decomposition, counted the number of vultures present, and noted additional evidence of feeding.

To determine if a relationship existed between the number of vultures at a carcass (group size) and carcass mass, we followed Houston (1988) and regressed group size (y) against carcass mass (x). We defined group size as the maximum number of vultures observed feeding, perched, and flying near a carcass. When follow-up visits were made to a carcass, we defined group size as the maximum number of vultures present during any single visit. Carcass mass was determined for each species from published sources (Schmidly 1994; Nowak 1999; Iverson et al. 2008; Hunter 2011; Castelló 2016). We used mean values for body mass; when only a range of values was given, the mid-point was used. Chi-square tests were used to compare group size during different stages of decomposition.

During September 2010 we conducted feeding trials on the Sul Ross State University (SRSU) Ranch adjacent to the main campus in Alpine. We deployed fresh (Stage 1 of Payne 1965) mammal carcasses with no discernible odor of decomposition (to us) obtained as roadkill (N = 3) or from animals prepared as study skins (N = 4) by SRSU students enrolled in a mammalogy class; the latter carcasses had the skin, head, and feet removed, but were otherwise intact and contained viscera. Carcasses were kept frozen until needed and then allowed to thaw overnight before being deployed the following day between 1000 and 1530 hr. Carcasses were placed in an open location (bare soil or closely cropped grass) visible to soaring vultures. We observed Turkey Vultures from a concealed location nearby and recorded the time elapsed between carcass deployment and arrival of the first vulture (defined as time when first vulture landed at carcass), maximum number of vultures present at the carcass, sequence of carcass processing, and the time required for vultures to consume the carcass. Statistical references are from Zar (1996). Mean values are presented as ± 1 SD and results were considered significant at $P \leq 0.05$.

RESULTS

We examined 59 carcasses that included representatives of 14 species of mammals and one reptile (Table 1). Mammals comprised the majority (96.6%) of our sample, and of these mule deer were the most common (Table 1). Thirty-nine carcasses (66.1%) were animals killed after colliding with vehicles, railroad mortalities accounted for four carcasses (6.7%), nine feral pigs were shot by landowners, and the cause of death of five domestic cattle and one domestic goat could

Table 1. Carrion consumed by Turkey Vultures in West Texas. Number of carcasses followed by percent of total (N= 59) for each species. Carcasses offered to Turkey Vultures during feeding trials (see text) are not included in this table. Body mass estimated from Schmidly (1994), Nowak (1999), Iverson et al. (2008), Hunter (2011), and Castelló (2016).

Species	Carcasses (%)	Body mass (kg)
MAMMALS		
Mule deer (*Odocoileus hemionus*)	18 (30.5)	79.5
White-tailed deer (*Odocoileus virginianus*)	2 (3.3)	40.0
Collared peccary (*Pecari tajacu*)	5 (8.4)	19.0
Domestic cattle (*Bos taurus*)	5 (8.4)	725.0
Domestic goat (*Capra hircus*)	1 (1.7)	66.5
Feral pig (*Sus scrofa*)	9 (1.5)	175.0
Coyote (*Canis latrans*)	1 (1.7)	17.0
Raccoon (*Procyon lotor*)	6 (10.1)	8.0
Striped skunk (*Mephitis mephitis*)	2 (3.3)	4.0
Gray fox (*Urocyon cinereoargenteus*)	3 (5.0)	4.0
Domestic cat (*Felis catus*)	2 (3.3)	1.8
Desert cottontail (*Sylvilagus audubonii*)	1 (1.7)	0.9
Rock squirrel (*Spermophilus variegatus*)	1 (1.7)	0.7
Porcupine (*Erethizon dorsatum*)	1 (1.7)	8.0
REPTILES		
Gopher snake (*Pituophis catenifer*)	2 (3.3)	0.4

not be determined. A gray fox carcass found at Elephant Mountain Wildlife Management Area was attributed to predation. We found a dead Turkey Vulture a short distance from the fox, apparently killed by the same predator. Feathers and other sign at the site suggested the vulture was killed while feeding on the fox and then transported about 5 m away by the predator. There was no sign of feeding by the predator on the vulture carcass.

We observed 254 Turkey Vultures at 39 carcasses and noted signs of feeding activity at 20 additional carcasses where no vultures were observed. Follow-up visits were made to 16 carcasses one to five days after discovery. We assessed most carcasses where vultures were observed as fresh or bloated, with only a few in the active decay and advanced decay stages of decomposition (Table 2). Insect larvae were present in carcasses during the active and advanced decay stages of decomposition. The number of vultures present on a carcass was not independent of the stage of decomposition ($\chi^2 = 19.1$, df = 3, $P < 0.05$). Vultures were more likely to be present on fresh and bloated carcasses when compared to those in the active and advanced stages of decay ($\chi^2 = 46.6$, df = 1, $P < 0.001$), and more likely to be present on bloated than fresh carcasses ($\chi^2 = 18.7$, df = 1, $P < 0.05$). We observed vultures consuming putrid viscera and muscle tissue from carcasses up to five days after the animal died. Additionally, we observed two vultures feeding on the dry remains of old (> 30 days) carcasses, and two vultures consuming meat scraps among discarded household waste at a public picnic area. The number of vultures feeding on a single carcass (group size) ranged from 1 to 20 and varied considerably (mean ± 1 SD = 6.1 ± 5.1 birds). We found a highly variable, yet significant positive correlation between group size and carcass mass ($R = 0.35$; $P < 0.05$; Fig. 1).

Turkey Vultures processed large and small to medium-sized carcasses differently. On carcasses of large (> 15 kg) road-killed mammals, vultures initiated feeding wherever the skin was torn and underlying muscle or viscera exposed. Vultures usually accessed the interior of larger carcasses (mule deer, collared peccary) through the abdomen,

Table 2. Turkey Vultures observed on carcasses during different stages of decomposition. Percent of total in parentheses. Stage of decomposition according to Payne (1965).

Stage of decomposition	Carcasses	Vultures
Fresh	22 (48.8)	95 (37.4)
Bloated	12 (26.6)	98 (38.5)
Active decay	10 (22.2)	54 (21.2)
Advanced decay	1 (2.2)	7 (2.7)
Total	45	254

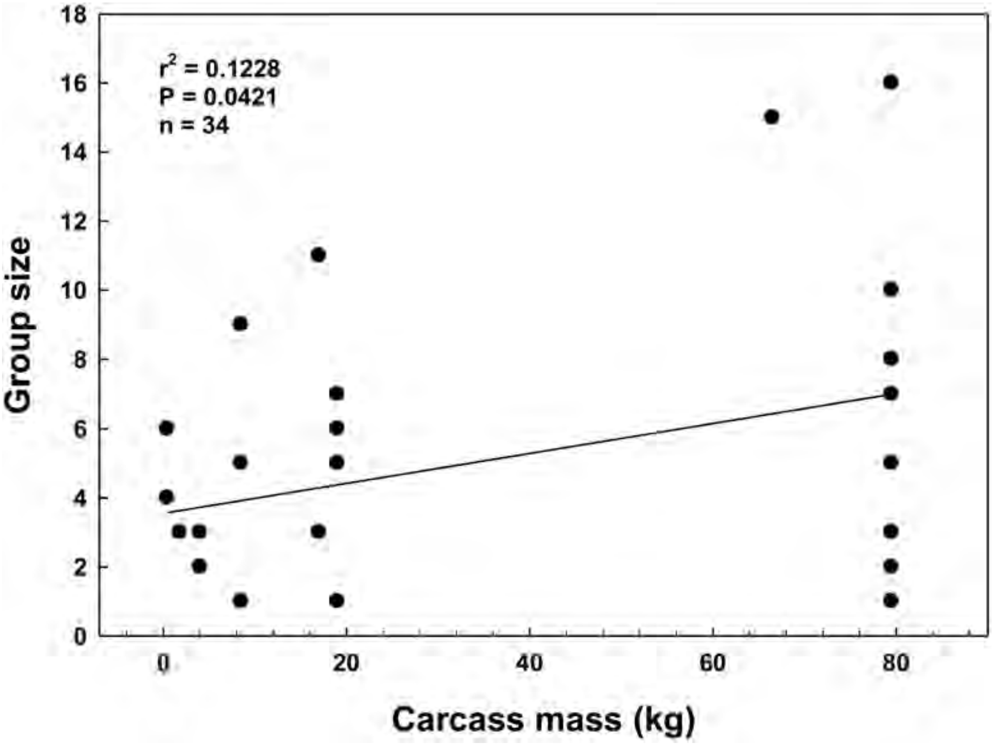


Figure 1. The relationship between the maximum number of Turkey Vultures present at a carcass (group size) and estimated carcass mass (does not include group sizes observed during feeding trials). Domestic cattle (with vulture group sizes of 18 and 20) were not included in this analysis, as the relatively large value for cattle body mass (725 kg) skewed the data. However, when domestic cattle were included, a stronger positive relationship between group size and carcass mass was observed ($P < 0.0001$).

which typically ruptures during collisions with vehicles. The eyes, tongue, and anus, and on occasion lips and ears, were among the first tissues consumed on large mammals, followed by viscera and muscle. In one instance, we observed vultures consuming the velvet and soft, underlying bone of developing antlers on a fresh mule deer carcass (Fig. 2). Unless exposed by injuries, the thoracic cavity was accessed after penetrating the diaphragm; vultures then consumed the heart and lungs. In one instance the shattered skull of a collared peccary allowed vultures to consume the brain, although generally these tissues remained inaccessible. Vultures consumed muscle by feeding outwards after first gaining access to the interior of a large carcass. Because vultures cannot penetrate the skin of most mammals, a considerable amount of muscle (particularly the lower legs) frequently remained uneaten on large carcasses. If a large carcass was intact with no injuries to provide access to the interior, vultures waited up to three days until the abdomen ruptured due to the build-up of gases

Figure 2. Developing antlers consumed by Turkey Vultures from the fresh carcass of a road-killed mule deer. Vultures consumed the velvet and soft, underlying bone as well as the tongue and one eye.

or the carcass was opened by another scavenger (e.g., coyote).

Carcasses of smaller (< 10 kg) road-killed mammals were processed in the same manner as large mammals with vultures accessing flesh and viscera through injuries sustained during collisions. As in larger carcasses, entry was usually through the abdomen. We found one raccoon carcass with an intact abdomen; about 4 hours later vultures had torn a small hole (ca. 1 cm in diameter) in the abdominal wall and extracted the viscera. Vultures were able to tear through and strip the skin from cottontail rabbit and rock squirrel carcasses. After gaining entry to the interior of small carcasses, vultures consumed viscera and muscle, peeling back the skin and removing muscle from even small bones (Fig. 3). Skeletal elements were often disarticulated by feeding vultures and transported a considerable distance (> 10 m) from the carcass. We documented the consumption of two porcupine carcasses by vultures; entry to an intact carcass (feeding trials) was accomplished by tearing through the thin abdominal skin, while the interior of a road-killed porcupine was accessed through injuries that opened the thoracic cavity (Fig. 4).

We observed seven Chihuahuan Ravens (*Corvus cryptoleucus*) in association with feeding Turkey Vultures on four occasions. Three observations consisted of single ravens, but a group four ravens was observed together with 25 vultures at a dump containing pig carcasses. We also observed a flock of House Sparrows (*Passer domesticus*) feeding for about 30 min on the badly decomposed remains of a small pig after vultures had consumed most of the muscle and viscera. After the sparrows departed, we found numerous maggots among the fur and other debris at the carcass. Presumably the sparrows were consuming maggots, but may also have been eating bits of flesh adhering to the skeleton.

During feeding trials at the SRSU Ranch, we observed Turkey Vultures consuming cottontail rabbit (N = 4), black-tailed jackrabbit (N = 1), porcupine (N = 1), and coyote (N = 1) carcasses. Mean (± 1SD) arrival time for vultures at carcasses (N = 7) was 33 ± 28 minutes (range = 5 to 82 minutes), and the maximum number of vultures present at carcasses ranged from 3 to 25 (mean ± 1SD = 10.4 ± 8.1 vultures); the largest number was observed at the unskinned carcass of a cottontail rabbit. Intense intraspecific aggression was noted among feeding

Figure 3. Fresh raccoon carcass after processing by Turkey Vultures. Vultures have peeled back the skin, consumed most of the viscera, and stripped muscle from the bones.

groups. Feeding vultures plucked fur and stripped large pieces of skin from intact cottontail and black-tailed jackrabbit carcasses to access muscle and viscera, demonstrating that the integument offers little hindrance when feeding on these thin-skinned taxa. We followed the complete carcass consumption sequence for three cottontail rabbits (range = 8 to 19 minutes), one black-tailed jackrabbit (41 minutes), and one porcupine (192 minutes). In the latter case, frequent disturbances caused vultures to briefly abandon the carcass before returning to feed. Vultures terminated feeding at a skinned coyote carcass after 270 minutes of intermittent activity during which the viscera and muscles from the hindquarters and rib cage were consumed. We were unable to resume observations on the following day, but 48 hr later only the disarticulated skeleton remained.

DISCUSSION

Our observations suggest that mammal carrion, particularly mule deer, is the primary food resource used by Turkey Vultures in west Texas. Our results are consistent with most previous studies which indicate that mammal carrion is the dietary mainstay of Turkey Vultures, although birds, reptiles, amphibians, fish, invertebrates, and even plant matter are also consumed (Kirk and Mossman 1998 and references therein; Rhodes 2007; Platt and Rainwater 2009; Platt et al. 2014). Similar to our findings, Davis (1979) reported that wild and domestic ungulates accounted for 15-24% of the food consumed by Turkey Vultures in West Texas, although in a study near El Paso Thomaides et al. (1989) found ungulates comprised only a minor component (7.7%) of the diet, perhaps owing to their rarity in the area. In the Eastern United States, white-tailed deer and domestic ungulates are important food resources for Turkey Vultures (Paterson 1984; Yahner et al. 1986; Coleman and Frazer 1987; Rhodes 2007; Jennelle et al. 2009), and in northern Mexico domestic ungulates comprise a significant portion of the diet (Hiraldo

Figure 4. Porcupine carcass partially consumed by Turkey Vultures. Access to the quill-studded carcass was attained through injuries sustained when the porcupine was struck and killed by a vehicle.

et al. 1991; Estrella 1994). Ungulate carcasses are a particularly rich source of nutrition for both avian and mammalian scavengers (Allen et al. 2014).

Several analyses of vulture pellets found a relatively high incidence of small carrion (e.g., rodents, insectivores, reptiles, and insects) in the diet (Paterson 1984; Thomaides et al. 1989; Prior 1990; Hiraldo et al. 1991), leading some authors to suggest that Turkey Vultures favor small carrion as a means of avoiding competition with sympatric Black Vultures (Stewart 1978; Coleman and Fraser 1987; Wallace and Temple 1987; Houston 1988; Estrella 1994). Although undoubtedly available to foraging vultures, small carrion was notably absent from our study. This is probably due to the bias inherent in observational studies of foraging vultures; small carrion is rapidly ingested either whole or piecemeal (Kirk and Mossman 1998; Platt and Rainwater 2009) and consumption is therefore likely to escape detection by observers. Alternately, the extreme rarity of Black Vultures in our study area may have allowed Turkey Vultures to exploit large carrion that would otherwise be unavailable due to interference competition by the former. Given the constraints on observational studies, perhaps the best approach in future dietary studies would be to combine pellet analysis with direct observations of feeding vultures.

We are unaware of any previous reports documenting the consumption of developing deer antlers or porcupine carrion by Turkey Vultures. In addition to being a protein source for vultures, developing antlers are also rich in calcium and phosphorous (Rue 1997), minerals that are especially important for growth and reproduction in birds (Beasom and Pattee 1978; Richardson et al. 1986). Porcupines are cloaked in barbed, keratinized quills that can migrate after penetrating the body with potentially lethal consequences (Katzner et al. 2015; Platt et al. 2016). Nine species of birds are known to scavenge or prey on porcupines, and 39% of the documented interactions between porcupines and birds resulted in death of the latter (Katzner et al. 2015). Our observations suggest that in some cases Turkey Vultures minimize the risk associated with consuming porcupines by accessing the carcass interior through the abdomen where quills are sparse or absent. We also found that on occasion, Turkey Vultures will consume household waste (see also Coleman and Fraser 1987), although this behavior is more commonly observed among Black Vultures, which often congregate in large flocks at municipal dumps (Buckley 1999).

Wildlife-vehicle collisions appear to be important for provisioning Turkey Vultures in West Texas (our study) and elsewhere (Rapp 1943; Theil 1976; Kirk and Mossman 1998; Rhodes 2007). Indeed, Rabenold (1989) suggested the widespread availability of road-killed carrion was a key factor in the range expansion of Turkey Vultures across the Northeastern United States. While undoubtedly an important food resource, there may be important functional differences between carcasses of road-killed wildlife and those resulting from natural predation. First, when large mammals are killed by vehicles, the location and extent of injuries determine how much of the carcass will be available to feeding vultures. Because Turkey Vultures are unable to penetrate the skin of most mammals (Houston 1988; Buckley 1996; this study), in some cases significant portions of a road-killed carcass cannot be consumed. By comparison, predators open and usually only partially consume large mammal carcasses, thereby providing ready access to an abundant nutritional reward for vultures (Crabtree and Sheldon 1999; Wilmers et al. 2003; Allen et al. 2014). Conversely, small to medium-sized prey may be completely consumed by predators with little remaining for vultures to scavenge. As a further consideration, the many small bone fragments (a potential mineral source for vultures) produced when predators process a carcass (Richardson et al. 1986) are probably much reduced or absent in road-killed carrion. And while vultures rarely fall victim to predators attending a carcass (see below), those feeding on road-killed carrion are at great risk of being killed by vehicles (DeVault et al. 2014).

Our feeding trials indicate that Turkey Vultures can readily locate fresh carrion placed in an open location visible to soaring birds. Our results contrast markedly with previous studies that found Turkey Vultures experienced great difficulty in locating fresh carcasses concealed beneath a forest canopy or buried below-ground, presumably because fresh carrion emits few odors of decomposition (Houston 1986; Platt et al. 2015). Given the absence of olfactory signals, the rapid arrival of vultures at fresh carcasses during our trials strongly suggests visual cues were used to detect carrion. Because Turkey Vultures possess a keen sense of smell (but see Smith and Paselk 1986) that is used to detect odors emanating from decomposing carcasses, any role that visual cues might play in foraging behavior is usually assumed to be minimal (Owre and Northington 1961; Stager 1964; Houston 1986; Estrella 1994). Indeed, Houston (1988) concluded that carcass detection depends "entirely" on the olfactory abilities of Turkey Vultures. However, our observations suggest otherwise and we see no *a priori* reason to dismiss the importance of visual cues to foraging Turkey Vultures. Because Turkey Vultures compete with a large suite of terrestrial and avian scavengers (DeVault and Rhodes 2002; Jennelle et al. 2009), the development of multiple mechanisms to rapidly detect carrion would seem to be adaptive and favored by natural selection.

In keeping with other reports (Owre and Northington 1961; Houston 1986), we found that Turkey Vultures generally preferred carrion in the early stages of decomposition. This finding was not unexpected as the nutritive value of a carcass declines with time as microbes rapidly metabolize those compounds of value to vultures (e.g., proteins, lipids, and carbohydrates) and toxins produced by decomposition steadily accumulate (DeVault et al. 2003). That said, we observed significant numbers of vultures feeding on putrid carrion (usually large ungulates) infested with insect larvae. This is likely due to the fact that vultures frequently had a lengthy wait until the abdomen ruptured and access could be had to the interior of the carcass, and then several days might elapse before a large ungulate could be consumed. Turkey Vultures are apparently not deterred by the presence of insect larvae in carrion (Kirk and Mossman 1998), and according to Houston (1988) "seem to enjoy [eating] maggots".

Similar to our study, others (Davis 1979; Houston 1988; Buckley 1996) also found a positive relationship between the size of feeding groups and carcass mass. Interestingly, the groups of Turkey Vultures we recorded at carcasses (up to 25 during feeding trials) appear to be among the largest feeding aggregations yet reported. Feeding groups described in previous studies are usually composed of 5-10 birds (Stewart 1978; Davis 1979; Coleman and Fraser 1987; Houston 1986; Buckley 1996). The large feeding aggregations we observed could be due to the rarity of Black Vultures in our study area. Where the two are sympatric, Black Vultures follow Turkey Vultures to carrion, aggressively dominate the carcass and exclude the latter from feeding (Stewart 1978; Wallace and Temple 1987; Houston 1988; Buckley 1996).

Our discovery of a dead Turkey Vulture beside a gray fox carcass appears to be the first record of a Turkey Vulture killed by a predator at a carcass and one of the few reports of mortality attributable to predation. In general, predation of adult Turkey Vultures appears rare (Kirk and Mossman 1998) and the available reports concern birds killed at communal roosts or while nesting; eggs and hatchlings are also reportedly taken by predators (Bent 1937; Coleman and Fraser 1986; Boal 2015). Although we were unable to identify the predator responsible for our observation, known predators of Turkey Vultures in North America include Golden Eagles (*Aquila chrysaetos*), Bald Eagles (*Haliaeetus leucoephalus*), raccoons, Virginia opossums (*Didelphis virginiana*), and domestic dogs (*Canis familiaris*) (Bent 1937; Coleman and Fraser 1986; Boal 2015).

ACKNOWLEDGMENTS

Support for this project was provided by Research Enhancement Grants from Sul Ross State University to SGP. Additional support for SGP was provided by the Wildlife Conservation Society-Myanmar Program. We thank Lewis Medlock for field assistance and bringing several carcasses to our attention, Deb Levinson and the library staff at Wildlife Conservation Society for obtaining a number of obscure, yet invaluable literature references, and Phil Wilkinson for preparing the photographs used in this manuscript.

LITERATURE CITED

Allen, M. L., L. M. Elbroch, C. C. Wilmers, and H. V. Wittmer. 2014. Trophic facilitation or limitation? Comparative effects of pumas and black bears on the scavenger community. PLOS ONE 9:e10227.

Barton P. S., S. A. Cunningham, D. B. Lindenmayer, A. D. Manning. 2013. The role of carrion in maintaining biodiversity and ecological processes in terrestrial ecosystems. Oecologia 171: 761–772.

Beasom, S. L. and O. H. Pattee. 1978. Utilization of snails by Rio Grande Turkey hens. Journal of Wildlife Management 42:916-919.

Bent, A. C. 1937. The Life Histories of North American Birds of Prey. Part 1. United States National Museum, Washington, District of Columbia.

Boal, C. 2015. Golden Eagle predation of an adult Turkey Vulture. Bulletin of the Texas Ornithological Society 48:53-55.

Buckley, N. J. 1996. Food finding and the influence of information, local enhancement, and communal roosting on foraging success of North American vultures. Auk 113:473-488.

Buckley, N. J. 1999. Black Vulture Vulture (*Coragyps atratus*). The Birds of North America Online: http://bna.birds.cornell.edu/bna/species/411 (accessed 1 November 2016).

Castelló, J. R. 2016. Bovids of the World: antelopes, gazelles, cattle, goats, sheep, and relatives. Princeton University Press, Princeton, New Jersey.

Coleman, J. S. and J. D. Fraser. 1986. Predation on Black and Turkey Vultures. Wilson Bulletin 98:600-601.

Coleman, J. S. and J. D. Fraser. 1987. Food habits of Black and Turkey Vultures in Pennsylvania and Maryland. Journal of Wildlife Management 51:733-739.

Crabtree, R. L. and J. W. Sheldon. 1999. The ecological role of coyotes on Yellowstone's northern range. Yellowstone Science 7:15-23.

Davis, D. 1979. Behavior of a breeding population of Turkey Vultures in West Texas. Ph.D. Dissertation, Colorado State University, Fort Collins, Colorado.

DeVault, T. L., B. F. Blackwell, T. W. Seamans, S. L. Lima, and E. Fernández-Juric. 2014. Effects of vehicle speed on flight initiation by Turkey Vultures: Implications for bird-vehicle collisions. PLOS One 9:e87944.

DeVault, T. L., B. D. Reinhart, I. L. Brisbin, Jr., and O. E. Rhodes, Jr. 2005. Flight behavior of Black and Turkey vultures: Implications for reducing bird-aircraft collisions. Journal of Wildlife Management 69:601-608.

DeVault, T. L. and O. J. Rhodes, Jr. 2002. Identification of vertebrate scavengers of small mammal carcasses in a forested landscape. Acta Theriologica 47:185-192.

DeVault, T. L., O. E. Rhodes, Jr., and J. A. Shivik. 2003. Scavenging by vertebrates: behavioral, ecological, and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. Oikos 102:225-234.

Estrella, R. R. 1994. Group size and flight altitude of Turkey Vultures in two habitats in Mexico. Wilson Bulletin 106:749-752.

Hiraldo, F., M. Delibes, and J. A. Donazar. 1991. Comparison of diets of Turkey Vultures in three regions of northern Mexico. Journal of Field Ornithology 62:319-324.

Houston, D. C. 1986. Scavenging efficiency of Turkey Vultures in tropical forest. Condor 88:318-323.

Houston, D. C. 1988. Competition for food in neotropical vultures in tropical forest. Ibis 402-417.

Hunter, L. 2011. Carnivores of the World. Princeton University Press, Princeton, New Jersey.

Iverson, J. B., C. A. Young, and T. S. Akre. 2008. Body size and growth in the bullsnake (*Pituophis catenifer* *sayi*) in the Nebraska sandhills. Journal of Herpetology 42:501-507.

Jennelle, C. S., M. D. Samuel, C. A. Nolden, and E.A. Berkley. 2009. Deer carcass decomposition and potential scavenger exposure to chronic wasting disease. Journal of Wildlife Management 73:655-662.

Katzner, T., T. A. Miller, J. Rodrigue, and S. Shaffer. 2015. A most dangerous game: death and injury to birds from porcupine quills. Wilson Journal of Ornithology 127:102–108.

Kirk, D. A. and M. J. Mossman. 1998. Turkey Vulture (*Cathartes aura*). The Birds of North America Online: http://bna.birds.cornell.edu/bna/species/339 (accessed 31 October 2016).

Leithead, H. 1959. Runoff in relation to range conditions in the Big Bend-Davis Mountains section of Texas. Journal of Range Management 12:83–87.

Nowak, R. M. 1999. Walker's mammals of the World. Volume 2. 6th Edition. The Johns Hopkins University Press, Baltimore, Maryland.

Olson, Z. H., J. C. Beasley, and O. E. Rhodes, Jr. 2016. Carcass type affects local scavenger guilds more than habitat connectivity. PLOS ONE 11(2):e0147798.

Owre, O.T. and P.O. Northington. 1961. Indication of the sense of smell in the Turkey Vulture, *Cathartes aura* (Linnaeus) from feeding tests. American Midland Naturalist 66:200-205.

Paterson, R. L., Jr. 1984. High incidence of plant material and small mammals in the autumn diet of Turkey Vultures in Virginia. Wilson Bulletin 96:467-469.

Payne, J.A. 1965. A summer carrion study of the baby pig *Sus scrofa* Linnaeus. Ecology 46:592-602.

Peterson, J. and B. R. Zimmer. 1998. Birds of the Trans-Pecos. University of Texas Press, Austin.

Platt, S. G., P. Charruau, and T. R. Rainwater. 2014. Scavenging of crocodile eggs by vultures (*Cathartes aura* and *Coragyps atratus*) in Quintana Roo, Mexico. Bulletin of the Texas Ornithological Society 47:37-40.

Platt, S. G., T. Gukian, R. E. Meraz, C. M. Ritzi, and T. R. Rainwater. 2015. Exhumation of buried mammal carrion by Turkey Vultures. Journal of Raptor Research 49:518-520.

Platt, S. G. and T. R. Rainwater. 2009. Noteworthy observations of foraging Turkey Vultures. Wilson Journal of Ornithology 121:839-841.

Platt, S. G., T. R. Rainwater, and S.T. McMurry. 2016. Predatory interaction between a Morelet's Crocodile (*Crocodylus moreletii*) and a Mexican Hairy Porcupine (*Sphiggurus mexicanus*) in Belize. Mesoamerican Herpetology 3:474-477.

Powell, M. A. 1998. Trees and Shrubs of the Trans-Pecos and Adjacent Areas. University of Texas Press, Austin.

Prior, K. A. 1990. Vulture food habits in southern Ontario. Wilson Bulletin 102:707-710.

Rabenold, P. P. 1989. Black and Turkey Vultures expand their ranges northward. Eyas 2:11-15.

Rapp, W. F., Jr. 1943. Turkey Vulture feeding habits. 1943. Auk 60:95.

Rhodes, O. E., Jr. 2007. Diet of Black and Turkey Vultures in a forested landscape. Wilson Journal of Ornithology 11:267-270.

Richardson, P. R. K., P. J. Mundy, and I. Plug. 1986. Bone crushing carnivores and their significance to osteodystrophy in Griffon Vulture chicks. Journal of Zoology (London) 210:23-43.

Rosenberg, K. V., and R. J. Cooper. 1990. Approaches to avian diet analysis. Studies in Avian Biology 13:80–90.

Rue, L. L., III. 1997. The deer of North America. Lyons Press, Boulder, Colorado.

Ruxton, G. D. and D. C. Houston. 2004. Obligate vertebrate scavengers must be large soaring fliers. Journal of Theoretical Biology 228:431-436.

Schmidly, D. J. 1994. The mammals of Texas. Texas Parks and Wildlife Department, Austin.

Smith, S. A. and R. A. Paselk. 1986. Olfactory sensitivity of the Turkey Vulture (*Cathartes aura*) to three carrion-associated odorants. Auk 103:586-592.

Stager, K. E. 1964. The role of olfaction in food location by the Turkey Vulture (*Cathartes aura*). Los Angeles County Museum, Contributions in Science 81. Los Angeles, California.

Stewart, P. A. 1978. Behavioral interactions and niche separation in Black and Turkey Vultures. Living Bird 17:19-34.

Thiel, R. P. 1976. Activity patterns and food habits of southeastern Wisconsin Turkey Vultures. Passenger Pigeon 38:137-143.

Thomaides, C., R. Valdez, W. H. Reid, and R. J. Raitt. 1989. Food habits of Turkey Vultures in West Texas. Journal of Raptor Research 23:42-44.

Wallace, M. P., and S. A. Temple. 1987. Competitive interactions within and between species in a guild of avian scavengers. Auk 104:290-295.

Wenny, D. G., T. L. DeVault, M. D. Johnson, D. Kelly, C. H. Sekercioglu, D. F. Tomback, and C. J. Whelan. 2011. The need to quantify ecosystem services provided by birds. Auk 128:1-14.

Wilde, M., and S. Platt. 2011. A life-giving trail: documenting the environmental history of Alamito Creek. Journal of Big Bend Studies 23:85–106.

Wilmers, C. C., R. L. Crabtree, D. W. Smith, K. M. Murphy, and W. M. Getz. 2003. Trophic facilitation by introduced top predators: Grey wolf subsidies to scavengers in Yellowstone National Park. Journal of Animal Ecology 72:909-916.

Yahner, R. H., G. L. Storm, and A. L. Wright. 1986. Winter diets of vultures in southcentral Pennsylvania. Wilson Bulletin 98:157-160.

Zar, J. H. 1996. Biostatistical Analysis. Prentice-Hall, Saddle River, New Jersey.

INFLUENCE OF LIGHT AND TEMPERATURE ON BRIDGE USE BY SWALLOWS

Lorissa J. Di Giacomo, Jacqueline Hernandez, Ivan Castro-Arellano, and M. Clay Green[1]

Department of Biology, Texas State University-San Marcos, San Marcos, Texas 78666, USA

ABSTRACT.—Habitat parameters that affect survival and reproduction can be enhanced or degraded from human activities including disturbance and development. While the development of human-made structures can obviously degrade (e.g. loss of habitat) a species' ability to survive and reproduce, human structures can also promote population growth through a species use of these structures for basic life history requirements, such as nesting and roosting. Our study examined the seasonal use of 5 bridges in Central Texas by Cliff Swallows *Petrochelidon pyrrhonota* and Cave Swallows *Petrochelidon fulva* during the nesting season. Specifically, we investigated the influence of temperature and ambient light properties on nest site selection. For both years of this study (2013-2014), Cliff Swallows were present during our surveys; while the numbers were variable between years and among bridges, in general Cliff Swallows were the dominant species present. In contrast, Cave Swallows were only recorded at two of the five sites: B2-Plum Creek, and B5- Blanco State Park during both years. We examined temperature and light properties at three bridges (B2, B3, B5) and found no significant interaction between bridges and probe ($F = 0.901$, $P = 0.493$) for mean temperature (°C) but the three bridges did significantly differed in mean temperature with B2 significantly warmer than B3 and B5 ($F = 15.104$, $P < 0.001$). For mean light (Lux), we found a significant interaction between bridge and probe ($F = 63.75$, $P < 0.001$) with all bridges receiving less light within the interior spans then the outer spans and the bridges differing significantly in overall ambient light; in order of decreasing light: B3, B2 and B5. Cave Swallows were found only within the interior spans of bridges (i.e. darker areas) or on the northern side of bridges and at the two bridges (B2, B5) that received the less light. However, Cave Swallows did not appear to be influenced by temperature as Cave Swallows occupied the hottest (B3) and coolest (B5) bridges. Based on our results, it appears Cave Swallows are selecting bridge site that are relatively dark but not influenced as much by temperature at the nest site. Future studies are warranted to continue investigating the nest site selection of Cave Swallows as they continue to expand their range into the south western United States.

Avian species use different resources within a variety of habitat types for all stages of their life cycle including nesting, foraging, and overwintering. Specific habitats have certain parameters that may promote survival and reproduction while other habitats may negatively impact the species' life history. Habitat parameters that affect survival and reproduction can be enhanced or degraded from human activities including disturbance and development. The development of human-made structures can obviously degrade (e.g. loss of habitat) a species' ability to survive and reproduce but can also promote population growth through a species use of these structures for nesting and roosting. These structures could also remove environmental barriers that limit a species' range and potentially result in the ranges of closely related species overlapping (i.e. from allopatry to sympatry); this appears to be occurring in two closely-related swallow species, *Petrochelidon pyrrhonota* (Cliff Swallow) and *P. fulva* (Cave Swallow) in south and central Texas. Cave Swallows have expanded their range in Texas and into south Florida (Martin 1974, Kosciuch et al. 2006, Strickler and West 2011). Cave Swallows have incorporated culverts, bridges and parking garages as well as caves for nesting and roosting sites; the range of the Cliff Swallow has not changed but they have included

---

[1] Email: laygreen@txstate.edu

human-made structures to their nesting and roosting sites (Kosciuch et al. 2006, Holderby et al. 2009, Strickler and West 2011). The increase in overlap (contact zone) with these anthropogenic structures provides the opportunity to examine resource use, spatially and temporally, between two closely related species.

Seasonal use of these structures is of interest because temporal variation in use may affect spatial occupancy (i.e. early arrivals choose nesting locations) as well as nesting productivity and survival of offspring. The contact zone in south and central Texas between Cave and Cliff swallows is an ideal region to examine the colony interactions and potential differences in colonization of these sites (Holderby et al. 2009). While both species of swallow may select the same structure (i.e. bridge), species preferences for nest site selection could still result in spatial and/or temporal separation at the colony. In a study of mixed-species waterbird colonies, Pius and Leberg (2002) hypothesized that Black Skimmer (*Rhynchops niger*) might be attracted to Gull-billed Terns (*Sterna nilotica*) within mixed-species colonies and therefore may nest in close association with Gull-billed Terns (Pius and Leberg 2002). However, Pius and Leberg (2002) found Black Skimmers nested in higher proportion next to skimmer decoys as opposed to tern decoys, suggesting, while nesting mixed-species colonies, individuals still opt to nest closer to conspecifics versus other species.

The overall goal of our research was to examine spatial use of nesting swallows in human-made structures. Specifically, we examined 1) the presence and maximum potential number of nests at each bridge by both species of swallow and, 2) spatial location of individual nests within each nesting structure (i.e. bridge) based on thermal and ambient light properties.

METHODS

Study Species

Cave Swallow and Cliff Swallow, closely related species, are morphologically very similar, with the most noticeable difference in the forehead patch color and throat color; with both colors tan in Cave Swallows and white and chestnut in Cliff Swallows, respectively (Brown and Brown 1995, Strickler and West 2011). While sexes are difficult to distinguish, females have a brood patch and in Cliff Swallows, the male has a dark blue path at the base of its throat (Brown and Brown 1995).

Both species are insectivorous and have been documented aerial foraging together in mixed–species flocks; swallows have been observed in mixed species communities where they act like a single community in their calls and foraging (Brown and Brown 1995, Weaver and Brown 2005, Strickler and West 2011). The Cliff Swallow migrates to Central and South America for the winter. Cave Swallows migrate to South America but some Texas birds will winter in the southern portion of Texas (Holderby et al. 2009). The Cliff Swallow's breeding range is from Alaska down to Baja California and Mexico and east into Connecticut (Brown and Brown 1995). The wintering range is from Brazil down into Paraguay (Brown and Brown 1995). The Cave Swallow's breeding range is from N.E. New Mexico into West and Central Texas down into Mexico. They also breed in Southern Florida, Great Antilles (Strickler and West 2011). The wintering range is similar to the breeding range they migrate south toward the borders of New Mexico, Texas (Strickler and West 2011). In southern Florida they migrate to the Caribbean Islands (Strickler and West 2011). In central Texas they have been documented to winter in their same breeding range (Strickler and West 2011).

During the breeding season both species will make nests from mud. They form the nests by adding mud with their beaks to the substrate (human-made or natural). Both species will use pre-existing nests and repair them if needed as long as the old nests have minimal to no swallow bugs (Brown and Brown 1996). They will line the nests with dry algae and plant material like grasses and cotton (Brown and Brown 1995, Strickler and West 2011). Swallows typically begin nesting in March/April and breeding season extends to as late as August (Brown and Brown 1995, Strickler and West 2011).

Study Sites

We surveyed swallow nesting at five different sites, all bridges, in the central region of Texas (Fig. 1). The selected study sites were examined a prior to project initiation for the presence of swallow nest substrate. The varying heights of the bridges at the study sites were taken in to account due to height restrictions potentially limiting the ability to count nests and mount environmental probes.

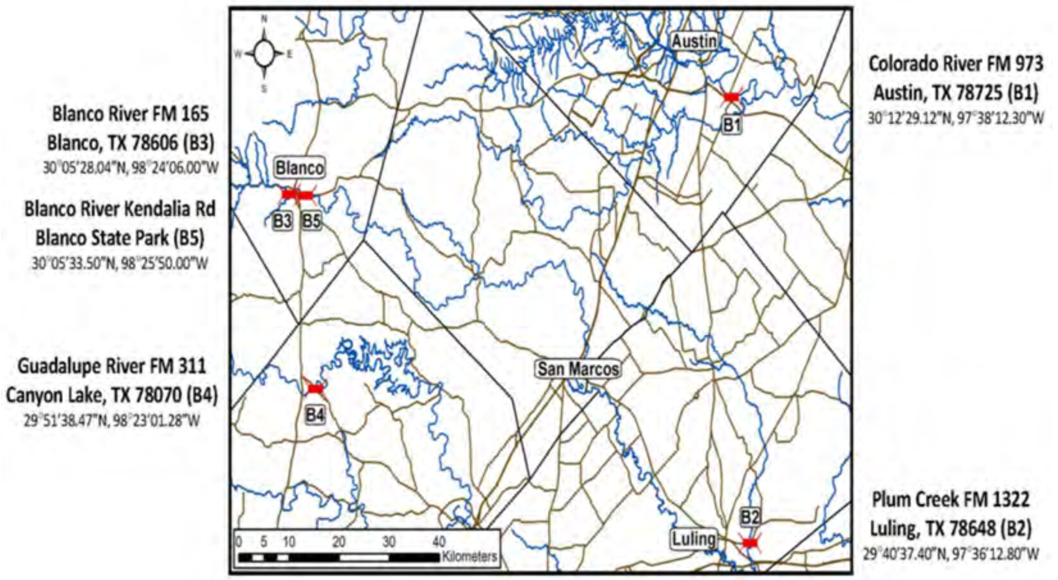


Figure 1. Locations of the 5 bridges surveyed weekly for the presence and maximum number of completed nests of swallow (*Petrochelidon* sp.) nesting colonies in Texas, 2013-2014.

Bridge height would affect distance between nests and water and therefore potentially affect species occupancy and number of birds. The height of the bridge over the bodies of water, in decreasing height, were as follows: Guadalupe River site 4 (B4) at 14.17 m, Colorado River bridge site 1 (B1) at 7.10 m, Plum Creek site 2 (B2) at 7.47 m, Blanco bridge site 3(B3) at 6.89 m and the Blanco State Park site 5 (B5) is the shortest at 4.94 m.

Bridge surveys

We observed and documented the arrival, placement and the numbers of active nests at each site on a weekly basis from February–August or until breeding swallows were no longer present for two years, 2013 and 2014. The nest occupancy by species was assessed through direct observation (e.g. spotting scope/binoculars). An active nest (complete nest) was defined as a nest with presence of swallows and/or signs of recently added materials as swallows often reuse nests from previous seasons but add material (usually mud) prior to nest initiation (Brown and Brown 1995, 1996). There are similarities between a complete Cave Swallow and a partially complete Cliff Swallow so to be able to tell the difference; presence of Cave Swallows had to be documented by sight or sound to confirm presence. In addition, photos were taken prior to arrival of swallows and then again periodically throughout the breeding season to document spatial and temporal changes in colony size and count complete nests.

Effects of temperature and ambient light

In 2014, we examined potential thermal and ambient light differences within and between study sites (B2- Plum Creek, B3- Blanco Bridge, B5- Blanco State Park) that may influence spatial segregation of Cave and Cliff Swallows nests; these bridges sites were chosen because of the confirmed presence of both species. We used Hobo® temperature/light data probe loggers (hereafter, probes), placing four probes at each site, two within the interior spans of bridges and two along the exterior spans of the bridge. All probes were placed along spans that contained nesting swallows. The probes were installed during March 2014 and taken down in September 2014. The bridges have an east-west orientation so the probes were labeled as follows: North outer, North interior, South interior and South outer. We compared ambient temperature and light measurements between bridges and within bridges, between spans, with the occurrence of Cliff and Cave Swallow nests to determine if light and/or temperature influences nest placement by each species. We conducted a multivariate analysis of variance (MANOVA) to examine differences in ambient light and temperature both within each bridge (between spans) and between bridges.

RESULTS

Bridge Surveys

For both years of this study, Cliff and Cave Swallows were present during our surveys; while the numbers were variable between years and among bridges, in general Cliff Swallows were present at all bridges and were the dominant species (between 97.3 and 100 percent of nests occupied) nesting in all of the five bridges. In contrast, Cave Swallow nests (presence of birds confirmed) were only recorded at two site B2-Plum Creek and B5-Blanco State Park during both years; at these sites nesting Cave Swallows still comprised a small number of individuals compared with Cliff Swallow individuals. The maximum number of complete nests at each bridge for 2013 and 2014, respectively, was as follows: 1) B1: 352, 230 nests, 2) B2: 221, 282 nests, 3) B3: 803, 779 nests, 4) B4: 301, 290 nests and, 5) B5: 123, 110 nests. For B2, there were 6 Cave Swallows nests (3 on North Interior, 3 on South Interior spans) counted in July 2013 and June 2014, respectively. For B3, although we observed Cave Swallows present at the bridge, no documented Cave Swallow nests were found in either years. For B5, we documented 3 Cave Swallow nests (2 on North Outer and, 1 on North Interior spans) in July 2013 and June 2014, respectively.

Effects of temperature and ambient light

We found no significant interaction between bridge and probe ($F = 0.901$, $P = 0.493$) for mean temperature (°C) but the three bridges (B2, B3, B5) did significantly differed in mean temperatures with B2 significantly warmer than B3 and B5 ($F = 15.104$, $P < 0.001$; Tables 1-3). For mean light (Lux), we found a significant interaction between bridges and probe ($F = 63.75$, $P < 0.001$) with all bridges receiving less light within interior spans then the outer spans and the bridges differing significantly in overall ambient light; in order of decreasing light: B3, B2,and B5.

DISCUSSION

Our surveys represent only a snapshot of nesting activity at each bridge site as birds are rarely at a nest for any lengthy period of time, often making repeated trips to and from the nest throughout the day to feed themselves, their mate and/or young (Brown and Brown 1996). When birds are present for any lengthy period of time, it is typically for incubation. However, when the female is incubating, she may be difficult to observe because the chimney shaped nest precludes internal observation of nest contents. The number of complete nests represents the total number of nests that were observed and classified as completely built. However, a complete nest does not necessarily represent an active nest but indicates the potential for nesting to occur. Nests and incomplete nests (e.g. partial nests) are present year round at sites and are likely re-used annually by nesting swallow pairs (Brown and Brown 1996). Our nest surveys provide an index of maximum possible nesting activity (i.e. complete nests). However, it should be noted that our estimated bird surveys do not suggest maximum colony sizes was reached during either nesting season (Di Giacomo 2015).

Our examination of the influence of light and temperature on spatial segregation of nesting swallows revealed that differences did exist between bridges (temperature) and between and within bridges (light). For temperature, B2 (Plum Creek) was surprisingly the warmest of the three measured bridges, surprisingly in that the bridge was more vegetated (Orsak 2014) and had greater average discharge of water underneath it than B3 or B5. Intuitively, the interior and outer spans differed significantly in light received with the interior portion of all bridges being darker. This is reasonable because the sun will hit the outer portion of the bridge for longer periods of time then the inner. The bridges did differ in light received, which corresponded with bridge height; in decreasing order of light and bridge height: B3, B2, and B5.

Cave Swallows were found only within the interior spans of bridges (B2, B5) or the northern side of the bridge (B5); both bridges that received less light. However, Cave Swallows did not appear to be influenced by temperature as Cave Swallows occupied the hottest (B2) and coolest (B5) of the three bridges. Based on our results, it appears Cave Swallows are selecting bridge site that are relatively dark but appear not to be influenced as much by temperature at the nest site.

Future studies are warranted to continue investigating the nest site selection of Cave Swallows as they continue to expand their range

Table 1: Comparison of mean temperature (°C) and mean ambient light (Lux) between the three study sites (i.e. bridges) and between probes, 2014. Bridges were as follows: B2, Plum Creek (29° 40’ 37.40” N, 97° 36’ 12.80” W), B3, Blanco River (30° 05’ 28.04” N, 98° 24’ 06.00” W), and B5, Blanco State Park (30° 05’ 33.50” N, 98° 25’ 50.00” W). All bridges were oriented approximately along east-west axis, therefore probes were position on outer and interior spans, north and south facing.

	Mean Temp (°C)	Mean Light (Lux)
Bridge		
B2 – Plum Creek	26.34 ± 0.22 A[1]	3374.40 ± 99.03 A
B3 – Blanco River	25.01 ± 0.21 B	3992.76 ± 128.83 B
B5 – Blanco State Park	24.83 ± 0.21 B	1951.13 ± 68.93 C
Probe		
North Outer	25.41 ± 0.25 A	5258.40 ± 89.89 A
North Interior	25.55 ± 0.25 A	1002.97 ± 24.70 B
South Interior	25.69 ± 0.25 A	878.63 ± 25.64 B
South Outer	24.92 ± 0.24 A	5288.41 ± 113.96 A

[1] Different letters within each subset (i.e. Mean Temperature at bridges) denotes significant differences based on Tukey’s HSD post-hoc test.

Table 2: Hobo® Probe results (mean ± (S.E.)) for the three bridges, 2014. Bridges were as follows: B2, Plum Creek (29° 40’ 37.40” N, 97° 36’ 12.80” W), B3, Blanco River (30° 05’ 28.04” N, 98° 24’ 06.00” W), and B5, Blanco State Park (30° 05’ 33.50” N, 98° 25’ 50.00” W).

	Bridge 2				Bridge 3				Bridge 5			
Probes	North Outer	North Inner	South Inner	South Outer	North Outer	North Inner	South Inner	South Outer	North Outer	North Inner	South Inner	South Outer
Variables												
Temp$_{min}$	21.23 (0.48)	23.42 (0.48)	23.58 (0.48)	20.74 (0.49)	20.25 (0.47)	21.18 (0.47)	21.25 (0.46)	19.61 (0.45)	20.57 (0.46)	21.27 (0.46)	21.27 (0.44)	20.32 (0.44)
Temp$_{max}$	31.79 (0.43)	30.34 (0.41)	30.38 (0.42)	31.08 (0.41)	31.22 (0.46)	28.92 (0.41)	28.97 (0.42)	30.5 (0.44)	31.18 (0.47)	28.75 (0.41)	28.86 (0.43)	28.98 (0.40)
Temp$_{mean}$	25.89 (0.43)	26.94 (0.44)	26.97 (0.44)	25.55 (0.42)	25.32 (0.43)	24.84 (0.44)	25.15 (0.43)	24.72 (0.41)	25.01 (0.43)	24.94 (0.42)	24.87 (0.42)	24.49 (0.40)
Temp$_{midpt}$	26.51 (0.44)	26.88 (0.44)	26.98 (0.44)	25.91 (0.43)	25.74 (0.44)	25.05 (0.43)	25.11 (0.42)	25.05 (0.42)	25.88 (0.44)	25.01 (0.42)	25.07 (0.42)	24.65 (0.40)
Light$_{min}$	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Light$_{max}$	23979 (407.4)	6276 (147.8)	3955 (74.4)	20707 (349.3)	22584 (386.6)	4208 (80.4)	4413 (84.1)	28046 (540.0)	16844 (523.9)	2419 (255.3)	3072 (199.4)	13755 (327.5)
Light$_{mean}$	5800 (139.2)	1371 (36.6)	1039 (48.6)	5298 (120.9)	6177 (160.4)	1223 (27.2)	1233 (29.2)	7338 (207.3)	3798 (85.6)	363 (10.5)	414 (9.6)	6200 (2981.5)
Light$_{midpt}$	11990 (203.7)	3138 (73.9)	1977 (37.2)	10353 (174.6)	11292 (193.3)	2104 (40.2)	2206 (42.1)	14023 (270.0)	8422 (261.9)	1210 (127.6)	1536 (99.7)	6877 (163.77)

Table 3: MANOVA results from probe data for the three bridges, 2014. Bridges were as follows: B2, Plum Creek (29° 40' 37.40" N, 97° 36'12.80" W), B3, Blanco River (30° 05' 28.04" N, 98° 24' 06.00" W), and B5, Blanco State Park (30° 05' 33.50" N, 98° 25' 50.00" W).

Source	Type III Sum of Squares	Df	Mean Square	F	Sig.
MEAN$_{temp}$					
Corrected Total	1187.10	11	107.918	3.744	P < 0.001
Intercept	1237161.81	1	1237161.81	42926.0	P < 0.001
Bridge	870.64	2	435.317	15.104	P < 0.001
Probe	159.45	3	53.149	1.844	P = 0.137
Bridge * Probe	155.82	6	25.970	0.901	P = 0.493
Error	54961.27	1907	28.821		
Total	1293305.83	1919			
MEAN$_{light}$					
Corrected Total	$1.103x\ 10^{10}$	11	1.003×10^{9}	617.59	P < 0.001
Intercept	1.853×10^{10}	1	1.853×10^{10}	11405.36	P < 0.001
Bridge	1.404×10^{9}	2	701961904	432.15	P < 0.001
Probe	9.009×10^{9}	3	3.003×10^{9}	1848.85	P < 0.001
Bridge * Probe	621328605	6	103443767	63.75	P < 0.001
Error	3.098×10^{9}	1907	1624339		
Total	3.265×10^{10}	1919			

into the southwestern United States (Kosciuch et al. 2006, Holderby et al. 2009). Kosciuch and Arnold (2003) first reported Cave Swallows using bridges but interestingly, all documented nests were in old Barn Swallow (*Hirundo rustica*) nests and did not report use of Cliff Swallow nests or nesting in close association with Cliff Swallows by Cave Swallows; our study is the first to report this novel nesting behavior. Future studies should investigate possible competition between Cliff and Cave Swallows and conduct a comparison of nesting success between the two congeneric species. Lastly, as Cave Swallows have only been recently documented to over-winter in Texas (Holderby et al. 2009), future studies that examine the influence of temperature and light on overwintering birds would also yield new insights into the continuing range expansion of Cave Swallows and potential impacts on Cliff Swallows.

ACKNOWLEDGMENTS

We thank Benjamin Orsak, Amanda Moore, Sara Harrod, Matthew Haverland and Rebekah Rylander for assistance in the field. We thank Floyd Weckerly for statistical advice. This manuscript was part of research project funded by Texas Department of Transportation (grant #0-6764).

LITERATURE CITED

Brown, C. R. and M. B. Brown. 1995. Cliff Swallow (*Petrochelidon pyrrhonota*), The Birds of North America Online. https://birdsna.org/Species-Account/bna/species/cliswa /introduction. (accessed 30 November 2016).

Brown, C. R. and M. B. Brown. 1996. Coloniality in the Cliff Swallow: The Effect of Group Size on Social Behavior. The University of Chicago Press, Chicago, Illinois.

Di Giacomo, L. J. 2015. Influence of light and temperature on the abundance of swallow nests. M.S. Thesis, Texas State University, San Marcos.

Holderby, Z. P., M. C. Green, and T. R. Simpson. 2009. The effects of body size and roost site on winter die-off of Cave Swallows. Bulletin of Texas Ornithological Society 42:82-86.

Kosciuch, K. L. and K. A. Arnold. 2003. Novel nesting behavior in Cave Swallows. The Wilson Bulletin 115(3): 347-348.

Kosciuch, K. L., C. G. Ormston, and K. A. Arnold. 2006. Breeding range expansion by Cave Swallows

(*Petrochelidon fulva*) in Texas. The Southwestern Naturalist 51: 203-209

MARTIN, R. F. 1974. Syntopic Culvert Nesting of Cave and Barn Swallows in Texas. The Auk 91:4:776-782

ORSAK, B. J. 2014. Impacts of Bridge Design and Land Cover Characteristics on Cliff Swallow Nesting. M.S. thesis, Texas State University-San Marcos, San Marcos.

PIUS, M. S. AND P. L. LEBERG. 2002. Experimental Assessment of the Influence of Gull-Billed Terns on Nest Site Choice of Black Skimmers. The Condor, 104(1):174-177.

STRICKLER, S. AND S. WEST. 2011. Cave Swallow (*Petrochelidon fulva*), The Birds of North America Online. https://birdsna.org/Species-Account/bna/species/claswa /introduction. (accessed 30 November 2016).

WEAVER, H .B. AND C. R. BROWN. 2005. Colony Size, Reproductive Success, and Colony Choice in Cave Swallows *Petrochelidon fulva*. Ibis 147 381-90.

SHORT COMMUNICATIONS

BLUE GROSBEAKS SUCCESSFULLY USE OLD ABANDONED BARN SWALLOW NEST

Ray C. Telfair II[1]

[1]11780 South Hill Creek Road, Whitehouse, Texas 75791

Blue Grosbeaks have not been reported to use abandoned nests of other species (Lowther and Ingold 2011); however, there is a report of a pair building a nest and fledging young in an Eastern Bluebird (*Sialis sialia*) box (Risch and Robinson 2006).

At Camp Tyler Outdoor School (15143 Camp Tyler Road, Whitehouse, Texas; 32°15'11.35"N, 95°11'24.46"W), on 6 April 2016, I approached an abandoned Barn Swallow (*Hirundo rustica*) nest that had been built and used for several years (but abandoned since 2012) on the upper right door frame at the back of the front porch of the vacant Farm Museum (Fig.1). The building is in an equine area of open pastures, various farm buildings, and scattered short to tall trees. A female Blue Grosbeak (*Guiraca caerulera*) flew from the nest which contained 3 eggs—pale blue, similar in color and size to those of the Eastern Bluebird (Baicich and Harrison 1997). On 13 April, the nest contained 4 eggs.

To avoid the possibility of desertion, the nest was not visited until 26 April when the chicks had hatched and were old enough to be banded (from youngest to oldest based on the degree of feather emergence). The chicks had large broad conical bills with bright yellow gape flanges and gray lower mandibles and the legs and feet were *2* dull

Figure 1. Location of abandoned Barn Swallow nest used by Blue Grosbeaks. Nest is on the molding above the right corner of the front porch door of the Farm Museum at Camp Tyler Outdoor School.

---

[1]Email: rctelfair@gmail.com

brownish–pink; the feathering was brownish mouse-gray (Baicich and Harrison 1997). On 4 May, the last two nestlings fledged. Apparently, there was no nest sanitation, i.e., nestling fecal sacs were not removed. The farm house wall adjacent to the nest and the nest cup and rim were coated in excrement deposited by the nestlings (Fig. 2). Nest sanitation for Blue Grosbeaks is not reported (Ingold 1993; Lowther and Ingold 2011).

Barn swallows nest on ledges, rafters, beams, and girders of buildings and under bridges or in culverts. There, they make an open shallow cup nest of mud pellets mixed with vegetable fibers and plant fragments sparsely lined with feathers and hair (Baicich and Harrison 1997, Brown and Brown 1999). In contrast, Blue Grosbeaks normally nest in a twig fork or among stems of shrubs, small trees, or tangles of vines often near open areas or roads. They build a cup shaped nest of thin twigs, bark strips, rootlets, dead leaves, corn husks, and, occasionally, cotton, rags, paper, string, cellophane, or cast snake skins lined with fine rootlets, tendrils, hair, and fine grasses (Ingold 1993, Baicich and Harrison 1997, Lowther and Ingold 2011). The swallow nest that was used by the grosbeaks contained no new lining material, only a few feathers and horsehair that had previously been gathered by the swallows. Although the nests of the two species are different in size and composition, the inside measurements are similar: swallow—7.5 cm inside diameter, 5.0 cm cup depth; grosbeak—6.4-7.6 cm inside diameter, 5.1 cm cup depth (Brown and Brown 1999a, 1999b; Ingold 1993, Lowther and Ingold 2011).

In summary, apparently, the use of an abandoned unmodified Barn Swallow nest by a pair of Blue Grosbeaks has not been previously documented. Also, this is an early nesting date—the first egg being laid on 4 April which is the earliest arrival time reported for the Smith County area (Dr. Peter Barnes, Tyler Audubon Society, pers. comm.; Northeast Texas Bird Checklist online, assessed 10 May 2016). However, the winter and spring were


05/04/2016 12:16

Figure 2. Coating of Blue Grosbeak nestling excrement deposits on the adjacent wall and in the Barn Swallow nest cup and along its rim.

mild and wet; so, if the migrating birds were in good breeding condition, it is quite possible that, since no time was used to build a nest, a nest could be found and the eggs laid only a few days prior to the date of average arrival. The successful fledging of the 4 chicks in an unusual but secure nest site is clearly beneficial; but, I do not have an explanation of the stimulus for this selection of this abnormal nest site.

I thank Camp Tyler Outdoor School for granting me permission to band birds on the property. I appreciate the expertise of Guy Sovia, computer consultant, for teaching me how to use the Microsoft Office Picture Manager.

LITERATURE CITED

Baicich, P. J. and C. J. O. Harrison. 1997. A guide to the nests, eggs, and nestlings, of North American birds, 2nd edition. Academic Press, San Diego, California.

Brown, C. R. and M. B. Brown. 1999a. Barn Swallow (*Hirundo rustica*). The Birds of North America, No. 452.

Brown, C. R. and M. B. Brown. 1999b. Barn Swallow (*Hirundo rustica*), The Birds of North America Online: *http://bna.birds.cornell.edu/bna/species/452* (assessed 4 May 2016).

Ingold, J. L. 1993. Blue Grosbeak (*Guiraca caerulea*). The Birds of North America, No. 79.

Lowther, P. E. and J. L. Ingold. 2011. Blue Grosbeak (*Passerina caerulea*). The Birds of North America Online: http://birds.cornell.edu/bna/species/79 (assessed 4 May 2016).

Risch, T. S. and T. J. Robinson. 2006. First observation of cavity nesting by a female Blue Grosbeak. Wilson Journal of Ornithology 118: 107-108.

WINTER NESTING OF WHITE-TAILED KITE IN SOUTH TEXAS

Michael Marsden[1], Michael Miller[2] and William S. Clark[3,4,5]

[1]61 Resaca Shores Blvd., San Benito, TX 78586
[2]100 East Wisteria, McAllen, TX 78504
[3]2301 South Whitehouse Circle, Harlingen, TX 78550

White-tailed Kites (*Elanus leucurus*) are fairly common breeding residents in the lower Rio Grande Valley of south Texas (http://txtbba.tamu.edu/species-accounts/white-tailed-kite/). According to Oberholser (1974), they breed in Texas from March to September, with egg dates from March 18 to August 21. The Birds of North America account for this kite reports young in the nest from mid-March into October (Dunk 1995).

MM & MM found a White-tailed Kite nest on 3 January 2015 about 8 Km east of Harlingen, Cameron County that had three well feathered nestlings. The nest was near the top of a large, mostly leafless hackberry tree (*Celtis occidentalis*) about 12m above the ground (Fig.1) and was located on the southern edge of the Los Palomitos Wildlife Management Area, Carricitos unit at the north end of Kornegay Road (Lat. 26°, 10.2' N, Long. 97°, 34.8' W). The chicks were still in the nest on a follow-up visits by WSC on 11 January and another by MMil on 20 January.

We estimate that eggs were laid in this nest in mid-November 2014, based on 30-32 days incubation and 4-5 weeks until fledging (Dunk 1995). Dunk (1995) reported the latest egg dates in North America as early July. However, Ferguson-Lees and Christie (2001) reported that they breed in nearby Nuevo Leon, Mexico from November onwards.

Lloyd Kiff (pers comm) found no winter breeding records for this kite from searches of several large raptor bibliographies, including Global Raptor Information Network (GRIN) and his personal one, RAPTOR.

This area has had more than average rainfall for the last two years, breaking a long drought. Rainfall recorded in nearby Harlingen was higher than average for November (6.45” vs 2.17”) and December 2014 (2.34” vs 1.82”) (http://nowdata.rcc-acis.org/bro/) As a result, there were many more microtine rodents, their preferred prey. Temperatures recorded in nearby Harlingen were also slightly higher than average in October, lower in November,

---

[4] Corresponding author

[5] Email: raptours@earthlink.net

Figure 1. White-tailed Kite nest with three chicks. January 2015. Photo by W. S. Clark.

and much higher in December, 2014 (http://www.srh.noaa.gov/bro/?n=2014event_charts).

Newton (1979) wrote that raptors breed when food is most readily available. The increase in prey and higher temperatures are the most likely reasons for this later than usual nesting.

ACKNOWLEDGMENTS

We thank Lloyd Kiff for searching for winter breeding records of this kite. Michael Fry and Craig Farquhar kindly provided helpful comments on an earlier draft.

LITERATURE CITED.

DUNK, J. R. 1995. White-tailed Kite (*Elanus leucurus*). The Birds of North America, No. 178.

FERGUSON-LEES, J., AND CHRISTIE, D. A. 2001. Raptors of the World. Christopher Helm, London.

NEWTON, I. 1979. Population Ecology of Raptors. Buteo Books. Vermillion, South Dakota.

OBERHOLSER, H. C. 1974. The bird life of Texas. (E. B. Kincaid, Jr. Editor) University of Texas Press, Austin.

GRAY HAWK FLEDGED FIVE JUVENILES IN SOUTH TEXAS

Richard Galindo[1] , Mary Gustafson[2], Roy Rodriguez[3] and William S. Clark[4,5]

[1]P.O. Box 5832 McAllen, TX 78501
[2]4211 Rio Grande Lane, Mission TX 78572
[3]Bentsen-Rio Grande Valley State Park, 2800 South Bentsen Palm Drive, Mission, TX 78572
[4]2301 South Whitehouse Circle, Harlingen, TX 78550

Gray Hawk (*Buteo plagiata*) is a Neotropical raptor that breeds from Costa Rica north to southern Arizona, New Mexico, and Texas (Bibles et al 2002). Clutch size varies, with a maximum of four eggs reported (Bibles et al 2002). Thus, it was of great interest when we learned of a Gray Hawk nest (Lat. 26° 10’ N, Long. 98° 23’W) in Bentsen Rio Grande State Park that had five nestlings. The nest with five chicks was reported to eBird by Cèdric Duhalde on 5 June 2015. MG and RR verified that there were indeed five nestlings shortly thereafter (Fig. 1). The stick nest was placed in an approximately 20 meter tall cedar elm (*Ulmus crassifolia*), and all chicks successfully fledged.

RG and family observed the nest and chicks for about 18 hours over several days before they fledged. They noted the absence of food begging vocalizations by the chicks, which were also observed casting aside partially-eaten prey for later consumption, as they were presumably full. No competition for food was observed. Hispid cotton rats (*Sigmodon hispidus*) and other rodent prey were the primary prey and were observed being skinned near the nest tree by the male (smaller) parent and then presented to the female (larger) for consumption or delivery to the nest.

WC contacted raptor researchers who have studied the breeding biology of this species and asked if they had ever recorded five eggs or chicks. All replied that four was the maximum they had found.

All five of the fledglings were seen flying with their parents later in the Park on 2, 10, and 16 Sept 2015, 4 Oct , and 11 Nov, 2016; and by themselves on 2, 6, and 21 Dec 2015 (John Kaye Pers. Comm.) and five juveniles seen flying together without their parents by RG and family on 31 Jan 2016 were presumed to be these juveniles (eBird checklist no.S27659332).

Figure 1. Five nestling Gray Hawks. All five fledged successfully. Bentsen Rio Grande Valley State Park, June 2015. (Photo Bill Supulski).

Several years of severe drought conditions in the Rio Grande Valley of South Texas were tempered by rainfall beginning in Sept 2013. Heavy fall and winter rains in 2014 officially ended this drought, and an El Niño southern-oscillation weather event produced record-breaking rainfall in the spring of 2015. National Weather Service records indicate that from Mar through May 2015, the greater area encompassing Bentsen Rio Grande State Park received more than 43 cm of rain, 300% of the normal rainfall for this three-month period. The annual rainfall reported at the nearby (11.5 km distant) McAllen International Airport for the previous year was almost double the yearly average (91.8 cm vs 55.6 cm average).

Periods of high rainfall have been correlated with pulses of seed and plant production. Such increases in forage have been noted to have a positive impact the population dynamics of rodents. (Davis and Schmidley 1997). In Texas, irruption of the hispid cotton rat population, among other rodent species, has been associated with high rainfall events and

---

[5]Corresponding Author Email: raptours@earthlink.net

explosive vegetation productivity (Davis and Schmidley 1997).

Newton (1979) reported larger clutch sizes in raptors with more food, both from supplemental feeding and from more natural prey, pointing out that "in all species large clutches are associated with rodent eating." While traditionally reported as favoring amphibian and reptilian prey, Gray Hawks also regularly take rodents. Varied species of amphibian and reptilian prey are also readily available in and around Bentsen Rio Grande State Park.

Bibles et al (2002) reported little of no information on the association of fledglings with their parents or siblings after fledging, primarily based on Arizona Gray Hawks. Contra to these, this species in southern Texas is non-migratory. That might help explain the observations of the five juveniles with their parents months after fledgling.

This is, to our knowledge, the first record of Gray Hawks fledging five juveniles.

ACKNOWLEDGMENTS

We thank Gia, Joaquin, Catarina and Gabriela Galindo for their enthusiastic assistance with field observations under difficult conditions and valuable contributions to this manuscript. We also thank Bill Supulski for permission to use his photo and John Kaye for sharing his observations.

LITERATURE CITED

Bibles, B. D., R. L. Glinski, and R. R. Johnson. 2002. Gray Hawk (*Astrina nitada*). The Birds of North America, no. 652.

Caister, L. 2009. Raptors, Rodents and Rare Weather: Managing Increased Migratory Raptor Populations at McConnell AFB, Kansas. 2009 Bird Strike North America Conference.

Davis, W.B., and D. J. Schmidley. 1997. The Mammals of Texas Online Edition. Texas Tech University. http://www.nsrl.ttu.edu/tmot1/sigmhisp.htm (accessed 14 December 2016).

Newton, I. 1979. Population Ecology of Raptors. Buteo Books. Vermillion, South Dakota.

Windberg, L. A. 1997. Population Trends and Habitat Associations of Rodents in Southern Texas. American Midland Naturalist 140: 153- 160.

EASTERN PHOEBES MOVE NEST LINING TO NEW NEST

Ray C. Telfair II[1]

[1]11780 South Hill Creek Road, Whitehouse, Texas 75791

Eastern Phoebes (*Sayornis phoebe*) renovate and renew old nests, both from year to year and for multiple broods within years; but, they have not been reported to move nest lining material (grass stems and hair) from one nest to another during a breeding season (Weeks 1994, 2011).

In 2013, a pair of Eastern Phoebes established a nest on the inside ledge at the top of a porch corner post (Fig. 1, Fig. 2) which is located at 11786 S. Hill Creek Road, Whitehouse, Texas (32°14'24.99"N, 95°12' 58.30"W). They successfully produced two broods. In 2014, their brood was eaten by a Texas Ratsnake (*Pantherophis* [*Elaphe*] *obsolatus*). The pair moved 9.7m to a nest site on the outer molding ledge at the left corner of the garage door (Fig. 3). There, they established a new nest and successfully raised the brood. In 2015, they returned to the original nest; but, the eggs were eaten by a ratsnake; so, they returned to the second nest and successfully produced a brood. In 2016, the birds returned to the first nest site which was in need of repair; removed lining from the second nest; and, took it to the first nest; but, there the eggs were also eaten by a ratsnake. They returned to the second nest site and lined it with recycled material from the first nest (Fig. 4). There, they were able to produce two successful broods.

The first nest site is in a shaded location near a wooded area where ratsnakes have good habitat and the corner post is easy for them to climb. The second nest site is in a sun exposed area (except for the nest which is shaded by the roof overhang. Also, there is a large amount of human activity involving vehicles moving in and out of the garage; and, much activity of small children playing nearby.

---

[1]Email: rctelfair@gmail.com

Figure 1. Locations of Eastern Phoebe nests at house porch corner post and left corner molding of garage door.

Figure 2. Original site of Eastern Phoebe nest on inside ledge at the top of a porch corner post.

The phoebes are habituated to these activities and display no avoidance.

I thank my neighbors—the Virginia Jackson Family—for informing me of the nesting activities of the phoebes and allowing me access to band the chicks. I appreciate the photographs (Figs.2 and 3) courtesy of Alan Byboth, Field Station Manager, Center for Biological Field Studies, Sam Houston

Figure 3. Site of second Eastern Phoebe nest on molding ledge at left corner of garage door.

Figure 4. Original site of Eastern Phoebe nest with lining material removed to reline the second nest.

State University, Huntsville, Texas. I also appreciate the expertise of Guy Sovia, computer consultant, for teaching me how to use the Microsoft Office Picture Manager.

LITERATURE CITED

Weeks, H. P., Jr. 1994. Eastern Phoebe (*Sayornis phoebe*). The Birds of North America, No. 94.

Weeks, H. P., Jr. 2011. Eastern Phoebe (*Sayornis phoebe*). The Birds of North America Online: *http://bna.birds.cornell.edu/bna/species/94* (accessed 7 July 2016).

FIRST VOUCHERED SPECIMEN OF SLATE-THROATED REDSTART (*MYIOBORUS MINIATUS*) FOR TEXAS WITH NOTES ON OTHER RECENT BREWSTER COUNTY SPECIMENS.

Heather L. Prestridge[1] and Gary Voelker

Biodiversity Research and Teaching Collections, Department of Wildlife and Fisheries Sciences, Texas A&M University, TAMU 2258, College Station, TX, USA.

A recent bird survey at Elephant Mountain Wildlife Management Area (EMWMA), Brewster County, Texas resulted in the first vouchered specimen of Slate-throated Redstart (*Myioborus miniatus*) for Texas, a new vouchered specimen record for White-eyed Vireo (*Vireo griseus* from Brewster County, and vouchers of an additional ten species for which modern specimens are lacking. The survey was conducted from 2-5 May 2016 in an effort to improve the availability of material for modern genetic studies and the general knowledge of avifauna of the area. A total of 68 species were observed and 39 species were collected and vouchered (Table 1). All specimens were deposited at the Biodiversity Research and Teaching Collections at Texas A&M University and data made accessible via the VertNet and iDigBio portals.

Elephant Mountain Wildlife Management Area (EMWMA), one of three management areas in the Trans-Pecos region of Texas, is maintained by Texas Parks and Wildlife Department as a wildlife research and demonstration area. It includes 23,147 acres of various habitat types including desert scrub, desert grassland, riparian zones, juniper-pinyon-oak woodlands, and deciduous canyon woodlands. Elephant Mountain, the highest and most prominent feature of the property, extends from northern to southern property boundaries and stands 6,225 feet above sea level. Two creeks run through the area providing drainage to the northeast (Chalk Draw) and west (Calamity Creek) of the mountain. We surveyed visually and via mist-netting. Our netting activities focused on the northern side of Elephant Mountain, taking advantage of a small patch of Cottonwood (*Populus deltoides*) located near Chalk Draw and the Chalk Trap House (30° 03’ 25.75” N, 103° 30’ 15.05” W, 1,317 m).

First vouchered specimen of Slate-throated Redstart (*Myioborus miniatus*) for Texas. Deposited at the Biodiversity Research and Teaching Collections at Texas A&M University, College Station.

We utilized the VertNet web portal to search for all vouchered specimens of birds from Brewster County held in collections. The VertNet resource provides biodiversity data from 330 collections globally, including all major and most minor collections in the United States. For sight records, we queried the

---

[1]E-mail: hlprestridge@tamu.edu

Table 1. Species collected from Elephant Mountain Wildlife Management Area, Brewster County, Texas May 2016. Vouchers are deposited at the Biodiversity Research and Teaching Collections, Department of Wildlife and Fisheries Sciences, Texas A&M University, College Station, Texas.

Trochilidae	Black-chinned Hummingbird (*Archilochus alexandri*)
Picidae	Ladder-backed Woodpecker (*Picoides scalaris*)
Tyrannidae	Hammond's Flycatcher (*Empidonax hammondii*)
Tyrannidae	Say's Phoebe (*Sayornis saya*)
Vireonidae	Bell's Vireo (*Vireo bellii*)
Vireonidae	Warbling Vireo (*Vireo gilvus*)
Vireonidae	White-eyed Vireo (*Vireo griseus*)
Fringillidae	House Finch (*Haemorhous mexicanus*)
Passerellidae	Rufous-crowned Sparrow (*Aimophila ruficeps*)
Passerellidae	Canyon Towhee (*Kieneria fusca*)
Passerellidae	Swamp Sparrow (*Melospiza georgiana*)
Passerellidae	Lincoln's Sparrow (*Melospiza lincolnii*)
Passerellidae	Green-tailed Towhee (*Pipilo chlorurus*)
Passerellidae	Chipping Sparrow (*Spizella passerina*)
Passerellidae	White-crowned Sparrow (*Zonotrichia leucophrys*)
Parulidae	Wilson's Warbler (*Cardellina pusilla*)
Parulidae	MacGillivray's Warbler (*Geothlypis tolmiei*)
Parulidae	Common Yellowthroat (*Geothlypis trichas*)
Parulidae	Black-and-white Warbler (*Mniotilta varia*)
Parulidae	Slate-throated Whitestart (*Myioborus miniatus*)
Parulidae	Northern Waterthrush (*Parkesia noveboracensis*)
Parulidae	Yellow Warbler (*Setophaga petechia*)
Parulidae	American Redstart (*Setophaga ruticilla*)
Icteridae	Yellow-breasted Chat (*Icteria virens*)
Icteridae	Bullock's Oriole (*Icterus bullockii*)
Icteridae	Orchard Oriole (*Icterus spurius*)
Icteridae	Brown-headed Cowbird (*Molothrus ater*)
Cardinalidae	Northern Cardinal (*Cardinalis cardinalis*)
Cardinalidae	Blue Grosbeak (*Passerina caerulea*)
Cardinalidae	Painted Bunting (*Passerina ciris*)
Cardinalidae	Varied Bunting (*Passerina versicolor*)
Cardinalidae	Western Tanager (*Piranga ludoviciana*)
Cardinalidae	Summer Tanager (*Piranga rubra*)
Remizidae	Verdin (*Auriparus flaviceps*)
Troglodytidae	Cactus Wren (*Campylorhynchus brunneicapillus*)
Troglodytidae	Bewick's Wren (*Thryomanes bewickii*)
Troglodytidae	House Wren (*Troglodytes aedon*)
Mimidae	Northern Mockingbird (*Mimus polyglottos*)
Turdidae	Hermit Thrush (*Catharus guttatus*)

popular citizen scientist sites eBird and iNaturalist for observations of species in Brewster County and solicited additional observations from the Texas Bird Records Committee.

NOTABLE SPECIMENS.

Slate-throated Redstart. The Slate-throated Redstart (*Myioborus miniatus*) specimen we collected is the first such record for Texas, and just the fifth vouchered specimen for the United States. The single female specimen (TCWC 23757, 8.2 g, ovary 5mm x 3mm smooth, no molt or fat, skull completely ossified) was collected by net in a cottonwood patch near Chalk Tank House, Elephant Mountain Wildlife Management Area, Brewster County, Texas on 2 May 2016. Other vouchers of the species from the United States are housed at the University of Michigan (UMMZ 208870, 16 April 1962, Lea County, New Mexico; UMMZ Birds Collection 2016) and University of Arizona (UAZ 12287 and 12848, 14 April 1976, Arizona, Cochise County, and UAZ 17957, 6 June 2005, Cochise County, Arizona; UAZ Bird Collection 2016).

In addition to our vouchered specimen, there are 13 accepted observational records of Slate-throated Redstart for Texas as recorded by the Texas Bird Records Committee (TBRC), a standing committee of the Texas Ornithological Society. This committee designated the species as a "review species" indicating that it has been sighted four or fewer times anywhere in Texas over a ten year average. Their records range from 1997 to 2015 and include six observations from Big Bend National Park in Brewster County (just south of EMWMA), and three from Davis Mountains Preserve in Jeff Davis County (just north of EMWMA). The remaining Texas sight records consist of single sightings from south Texas in Corpus Christi (Nueces County), Pharr (Hidalgo County), and South Padre Island (Cameron County), and one from Plains (Yoakum County) in northwest Texas (Texas Ornithological Society 2016).

Although annual reports of the TBRC are available on-line, TOS does not publish their observational data via any of the popular web portals that aggregate biodiversity data such as VertNet or iDigBio. However two of the records (Davis Mountains Preserve from 1997 and Big Bend National Park from 2006) do appear on iNaturalist (iNaturalist 2016). Two additional records for the species appear on eBird, yet these observations lack photographic evidence. The eBird records include one observation from 1990 in Boot Springs, Big Bend National Park and the other from Jeff Davis County at the Nature Conservancy Preserve in the Davis Mountains (eBird 2016).

White-eyed Vireo. One specimen of White-eyed Vireo (*Vireo griseus*, TCWC 23744) was collected during our survey of EMWMA on 4 May 2016, also in our mist net arrays at Chalk Tank House, and represents the first vouchered specimen of the species for Brewster County, Texas. This specimen likely represents a rare occurrence, as there are no other specimen records west of Uvalde County and there are few observations for Brewster County that appear on eBird (eight observations dating between 1983 and 2014).

Other records. In addition, ten of the species that we vouchered are the first specimens from the county

Table 2. Last vouchered specimens for species collected at Elephant Mountain Wildlife Management Area in May 2016 for which modern collections have been lacking.

Species	Last voucher date	Locality	Museum number
Warbling Vireo (*Vireo gilvus*)	5/27/1935	Alpine; 4 miles west ; Paradise Canyon	CM[1] P117452
Green-tailed Towhee (*Pipilo chlorurus*)	3/29/1958	Black Gap, 2300 ft.	TCWC[2] 6073
MacGillivray's Warbler (*Geothlypis tolmiei*)	5/12/1968	6 mi N, 2 mi W Alpine	ROM[3] 100560
Black-and-White Warbler (*Mniotilta varia*)	5/15/1932	Chisos Mountains, Boot Spring	UMMZ[4] 86288
Northern Waterthrush (*Parkesia noveboracensis*)	9/10/1956	Black Gap, 2069 ft.	TCWC 5991
Yellow Warbler (*Setophaga petechia*)	8/27/1955	Oak Cr. 4000 ft., Chisos Mts.	TCWC 5986
American Redstart (*Setophaga ruticilla*)	*/*/1975	Big Bend	NCSM[5] 5137
Western Tanager (*Piranga ludoviciana*)	5/1/1970	Black Gap Wildlife Management Area Headquarters.	PMNS[6] ORN006122
Summer Tanager (*Piranga rubra*)	8/22/1979	7 mi N Alpine	ROM 136584
House Wren (*Troglodytes aedon*)	10/3/1975	14 mi S, 2 mi W Fort Davis	ROM 124421

[1]CM, Carnegie Museum; [2]TCWC, Texas Cooperative Wildlife Collection; [3]ROM, Royal Ontario Museum; [4]UMMZ University of Michigan; [5]NCSM, North
Carolina State Museum; [6]PMNS, Perot Museum of Nature and Science.

in 25 years or more; for those species our modern collections represent the only tissues available for genetic work from Brewster County (Table 2).

The importance of modern specimens which are accessible to the scientific community at large cannot be understated. Primarily, these collections document diversity, its presence or absence in a given location, and the variation in that diversity across the landscape and through time. By the manner in which modern voucher specimens are preserved, data associated with them taken, and made readily accessible by repositories such as the BRTC these collections also provide an important resource from which broader scientific questions can be addressed. These questions can include genetic variation, host-parasite interactions, and disease ecology in and between wildlife and domestic animals (e.g., livestock).

ACKNOWLEDGMENTS

We thank Russell Martin, Mark Garrett, and Dewey Stockbridge of TPWD. Specimens were collected under SPR 0209-016 and USFWS Federal Bird Permit #MB205752-2. This is publication number 1540 of the Biodiversity Research and Teaching Collections at Texas A&M University.

LITERATURE CITED

eBird. 2016. eBird: An online database of bird distribution and abundance. eBird, Ithaca, New York. http://www.ebird.org. (accessed: 7 July 2016).

iNaturalist. 2016. iNaturalist: An on-line community of naturalists [web application]. iNaturalist California Academy of Natural Sciences, San Francisco, California. Available: http://www.inaturalist.org. (accessed: 7 July 2016).

Texas Ornithological Society. 2016. Texas Bird Records Committee Reports http://www.texasbirdrecordscommittee.org/home/reports/report-archives

UAZ Bird Collection. University of Arizona Museum of Natural History. http://ipt.vertnet.org:8080/ipt/resource.do?r=uaz_birds (accessed on 27 June 2016).

UMMZ Birds Collection. University of Michigan Museum of Zoology. http://ipt.vertnet.org:8080/ipt/resource.do?r=ummz_birds (accessed on 27 June 2016).

PREDATION ON TEXAS HORNED LIZARDS BY GREAT-TAILED GRACKLES IN THE HIGH PLAINS OF TEXAS

James D. Ray[1]

8500 Kemper Road, Canyon, TX 79015 USA

The Great-Tailed Grackle (*Quiscatus mexicanus*) was restricted in distribution in the United States prior to 1900 to extreme southern Texas (Wehtje 2001). By 1930, the bird had expanded its range to include coastal plains habitats of the Gulf Coast States of Florida and Texas, and the Southern Atlantic states of Delaware through Florida (G. S. LeBaron, Bird Source http://www.birdsource.org/Features/Grackles). In Texas the Great-Tailed Grackle could be found nesting as far north as the Panhandle and Dallas-Fort Worth by 1960 and statewide by the 1970s (G. S. LeBaron, Bird Source http://www.birdsource.org/Features/ Grackles; Davis 1975; Seyffert 2002). Explosive growth occurred after 1960, coinciding with human-induced habitat changes such as irrigation's influence on farming and lawn maintenance (Johnson and Peer 2001), provisioning of bird feeders (Fronimos et al. 2011), and establishment of livestock feedlots (USDA-NASS 2000). Great-Tailed Grackles require trees or cattail marshes for nesting (Johnson and Peer 2001) and the establishment of trees around human habitations in formerly treeless areas became sources of nesting structure (Arnold 2001). Their use of clumps and rows of introduced trees (particularly *Gleditsia triacantho* and *Robinia pseudoacacia*) in highway rest areas and roadside tree plantings distant from towns in High Plains habitats of the Panhandle has increased to the point that most now host nesting colonies of these birds (J. D. Ray, unpubl. data).

---

[1] E-mail: Corresponding author: jdraypuma@gmail.com

Species that have undergone such range extensions can evoke negative effects on species endemic to the invaded area (Gutierrez et al. 2007; Farrell et al. 2011). For example, Brown-Headed Cowbirds (*Molothrus ater*) have increased or spread into adjacent areas due to alterations of habitat (agriculture, fire suppression, anthropogenic structures [perch sites], etc.) and affected productivity of other birds including species of concern (Shaffer et al. 2003; Gutierrez et al. 2007).

I found no evidence in the literature of negative impacts of Great-Tailed Grackles on species of concern although I have observed two instances of their predation on Texas horned lizards (*Phrynosoma cornutum*) on the High Plains of the Texas Panhandle. This lizard is one of four species of horned lizards that occupy Texas and is classified as threatened by the state of Texas.

In the pm of June 1991, I observed a female Great-Tailed Grackle ~4 m away with a neonate Texas horned lizard in her beak, "beat" it three or four times against a garden hose lying on the ground in my backyard in Canyon, Randall Co. (34°58'38.01"N; 101°56'38.29"W). She then flew off with the horned lizard in a SE direction.

In the pm of June 2014, I observed a female Great-Tailed Grackle walking with a neonate Texas horned Lizard in her beak ~2 m off the shoulder (north) of the westbound access road of Interstate 40 (Carson Co.), 3-5m west of its intersection with FM 2373 (35°13'22.96"N; 101°32'35.80"W). The grackle took flight a few seconds after I came to a stop upon seeing the bird while making the right-hand turn on to the access road. Its flight path was towards a nesting colony of a dozen nests or more located in the roadside park (westbound side) 0.5 km to the SW/0.5 km W from FM 2373 overpass/I-40.

Great-Tailed Grackles opportunistically consume a wide range of food items including grains and seeds; fruits and berries; plant sprouts and tubers; aquatic, ground and aerial insects (including larvae); spiders; crustaceans; snails and slugs; amphibians (including tadpoles); reptiles; mice and shrews; birds and bird eggs (including eggs and young of conspecifics); small fish; and they scavenge carrion and dropped or discarded human foods (Davis and Arnold 1972; Johnson and Peer 2001; Fronimos et al. 2011). Scavenging for carrion on roadways is common for this species and this could have been the case with my June 2014 (roadside) observation, but the group of females (6-7) that accompanied the female that was carrying the horned lizard were all actively foraging in the grassy roadside (picking among the vegetation, making short dashes and hops after prey).

My observations are the first published for predation of Texas horned lizards by Great-Tailed Grackles, providing confirmation to an anecdotal mention as "suspected" in an informational bulletin produced by the Oklahoma Department of Wildlife Conservation, Wildlife Diversity Program (1996). I also could not find any published reports of predation on horned lizards by the similar, but coastally-restricted Boat-Tailed Grackle (Q. major), which was once considered conspecific (Dacosta et. al 2008).

The Texas horned lizard formerly occurred statewide, but is now mostly absent from east of a line that extends from Fort Worth through Austin and San Antonio to Corpus Christi (Price 1990). Declines are also apparent in Central Texas (Donaldson et al. 1994). Although it is not known what level of impact Great-Tailed Grackles have had on declines in Texas horned lizard populations, its potential contribution to local declines within foraging distances of Great-Tailed Grackle nesting and roosting sites should not be over-looked. This may be especially important as habitat modifications continue to favor the adaptable Great-Tailed Grackle and other predators in more rural areas where Texas horned lizard habitat remains.

ACKNOWLEDGMENTS

I am grateful to Clint Boal for his review of this note.

LITERATURE CITED

Arnold, K. A. 2001. Great-Tailed Grackle. The Texas Breeding Bird Atlas. Texas A&M University System, College Station and Corpus Christi, TX. http://txtbba.tamu.edu (accessed 12 May 2016).

Dacosta, J. M., W. Wehtje, and J. Klicka. 2009. Historic genetic restructuring paraphyly within the Great-Tailed Grackle. The Condor 110(1):170–177.

Davis, W. M. 1975. The Great-Tailed Grackle in Oklahoma. Bulletin of the Oklahoma Ornithological Society VIII(2):9-20.

Davis, W. R. and K. A. Arnold. 1972. Food habits of the Great-Tailed Grackle in Brazos County, Texas. Condor 74:439-446.

Donaldson, W., Price, A.H. and Morse, J. 1994. The current status and future prospects of the Texas horned lizard (*Phrynosoma cornutum*) in Texas. Texas Journal of Science 46(2): 97-113.

Farrell, S. L., M. L. Morrison, R. N. Wilkins, R. D. Slack, and A. J. Campomizzi. 2011. Brown-

Headed Cowbird parasitism on endangered species: relationships with neighboring avian species. Western North American Naturalist 70(4):474-482. https://www.researchgate.net/publication/41031175_Brown-headed_cowbird_parasitism_on_endangered_species_relationships_with_neighboring_avian_species. (accessed 17 May 2016).

Fronimos, A. B., J. T. Baccus, M. F. Small, and J. A. Veech. 2011. Use of urban bird feeders by White-Winged Doves and Great-Tailed Grackles in Texas. Bulletin of Texas Ornithological Society 44(1-2):34-40.

Gutierrez, R. J., M. Cody, S. Courtney, and A. B. Franklin. 2007. Invasion of Barred Owls and its potential effect on the Spotted Owl: a conservation conundrum. Biological Invasions 9:181-196. http://link.springer.com/article/10.1007%2Fs10530-006-9025-5 10.1007/s10530-006-9025-5. (accessed 16 May 2016).

Johnson, K., and B. D. Peer. 2001. Great-tailed Grackle (*Quiscalus mexicanus*), The Birds of North America Online: http://bna.birds.cornell.edu/bna/species/576. doi:10.2173/bna.576. (accessed 8 May 2016).

Oklahoma Department Of Wildlife Conservation, Wildlife Diversity Program. 1996. Oklahoma's most wanted: the Texas horned lizard. Bulletin 36024. U.S.A. Oklahoma Wildlife Diversity/U.S. Fish and Wildlife Service/Section 6 Endangered Species Act Grant. Oklahoma City.

Price, A. H. 1990. *Phrynosoma cornutum.* Catalogue of American Amphibians and Reptiles 469: 1-7.

Seyffert, K. 2002. Birds of the Texas Panhandle: their status, distribution, and history. Texas A&M University Press, College Station.

Shaffer, J. A., C. M. Goldade, M. F. Dinkins, D. H. Johnson, D. Lawrence, and B. R. Euliss. 2003. Brown-Headed Cowbirds in grasslands: their habitats, hosts, and response to management. USGS Northern Prairie Wildlife Research Center. Paper 158. http://digitalcommons.unl.edu/usgsnpwrc/158. (accessed 16 May 2016).

USDA-NASS. 2000. Quickstats. Cattle on Feed Report 1965-2000. http://www.usda.gov/NASS.

Wehtje, W. 2001. Range expansion of the Great-Tailed Grackle in western North America. Western Birds 32:141-143.

USING WING LENGTH TO AGE JUVENILE MONTEZUMA QUAIL

Jack Eitniear[1] and Terry Becherer[2,3]

[1]218 Conway Drive, San Antonio, Texas 78209
[2]2312 Spruce Cabin Road, Cresco, Pa. 18326

The Montezuma Quail *Cyrtonyx montezumae* inhabits the pine-oak forest, arid montane scrub, and grasslands throughout the mountains of Mexico. Its distribution reaches the United States only in southern Arizona, New Mexico, (Stromberg 2007) and western Texas (Baccus and Eitniear 2007, Eitniear 2014). Despite the significant amount of literature on the species we found no guide for aging Montezuma Quail chicks (Fig.1). The variability in body weight between individuals, due to yearly differences in the food supply (as well as the difference between wild and captive diets), precluded using weight as an accurate age indicator; therefore we relied instead on wing measurements (Smith and Cain 1984, Leopold 1939, Petrides and Nestler 1943).

METHODS

Thirty five randomly selected day old chicks, hatched from eight pairs of Montezuma Quail, held in captivity (for husbandry details see Eitniear and Becherer 2012) at Becherer's aviary in Pennsylvania, were selected. Wing length was measured by holding the wing perpendicular to the body in a frontal plane (Eck et al. 2012) Measurements were taken once weekly for 6 weeks to the nearest millimeter, with all 35 being measured on the last day of the week. Since we could not sex day old quail chicks all measurements were pooled. Adults were also measured to obtain an adult measurement.

RESULTS AND DISCUSSION

Montezuma Quail chicks reached 92% adult wing length by 6 weeks (Fig. 2). However, our average adult wing length of 129 mm is 6 mm greater than the adult measurement provided by Stromberg

---

[1] Email: jclintoneitniear@gmail.com

[3] Deceased

(2000): adult wing length of 123 mm. (n=54) for males[1] and 119 mm (n=30) for females[1.] The difference may be an artifact of multi-generational breeding in captivity.

Adult weigh is more variable, due to the reasons previously stated, and likely requires considerable more time especially considering that Scaled Quail (*Callipepla squamata*) chicks reach about 50 percent of adult body weight by 6 weeks, and by 13 weeks they are 90 percent of mature weight (Cain and Beasom 2011). Given that we fed a commercial diet, and exercise was limited, weight gain in captivity is likely more rapid than in the wild. Nevertheless, it is likely that 13 to 14 weeks are required to attain an adult weight. However, as was discovered by Smith and Cain (1984), large coefficients of variation preclude the use of weight as an accurate age criterion.

Figure 1. Day old Montezuma Quail.

At four to six weeks the black on the belly and top of leg appears, indicating chicks that were males. While we pooled all measurements it should be noted that the group contained 25% females; therefore wing length measurements are biased towards males. Stromberg (2000) indicated only a difference of 0.4 mm between sexes so our polling of both sexes likely made minimal difference.

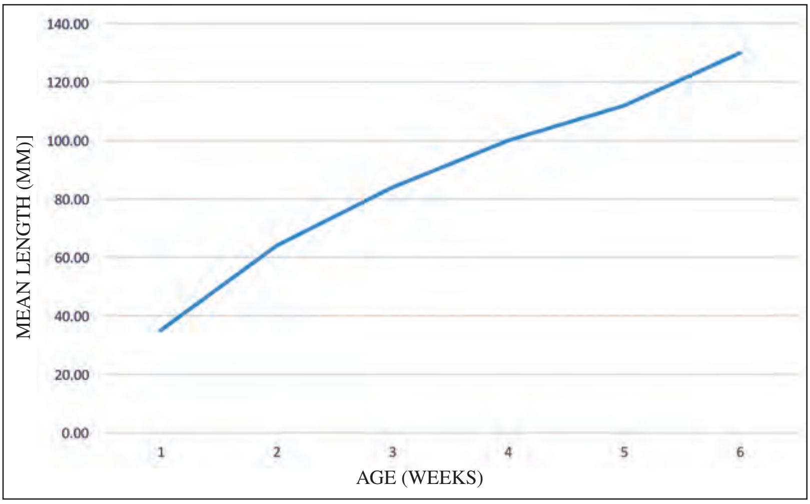


Figure 2. Mean weekly wing length (mm±1 SD)) for 35 Montezuma Quail from hatch to 6 weeks of age.

Replicating this study with wild caught birds is problematic however. Until the differences in adult wing length between our captive birds and museum skins can be explained, we suggest any use of our graph as the bases for aging should be done cautiously.

LITERATURE CITED

Stromberg, M. R. 2000. Montezuma Quail (Cyrtonyx montezumae), The Birds of North America Online: http://bna.birds.cornell.edu/bna/species/524 (accessed 05 December 2016)..

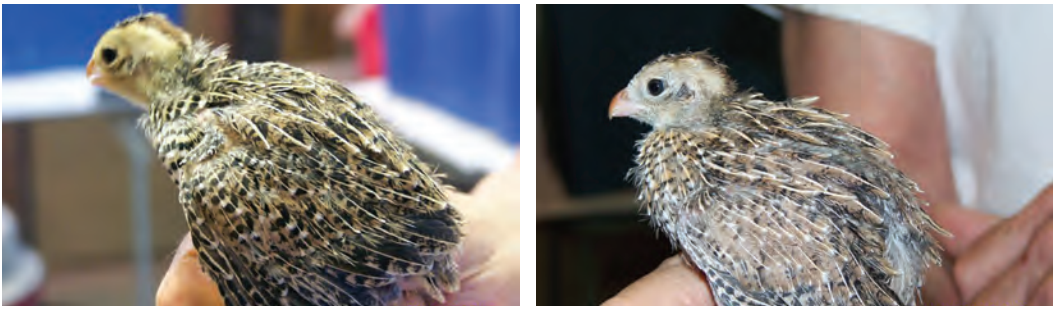
Baccus, J. T. and J. C. Eitniear. 2007 Quail of the Edwards Plateau, Pages 217-232 in Texas quails: ecology and management, (Leonard Brennan, Editor). Texas A&M University Press, College Station.

Cain, J. R. and S .L. Beasom 2011 A guide for aging Scaled Quail. Texas Agricultural Experimental Station http://baylor.agrilife.org/files/2011/06/Guide_For+_Aging_Scaled_Quail_1.pdf.

Eck, S., Fiebig, J, Fiedler, W., Heynen, I., Nicolai, B., Töpfer, T., Van Den Elzen, R. Winkler, R. and F. Woog 2011 Measuring Birds—Vögel vermessen.

---

[1] From Specimens at the Museum of Vertebrate Zoology, University of California Berkeley



2 weeks

3 weeks

4 weeks

5 weeks

6 weeks

Figure 3. Images of known age Montezuma Quail chicks. At four to six weeks the black on the belly and top of leg appears, indicating chicks that were males.

Herausgeben von der Deutschen Ornithologen-Gesellschaft. Christ Media Natur, Minden, 122 Seiten. Germany.

Eitniear, J. C. and T. Becherer. 2012. Dietary considerations when breeding Montezuma Quail *Cyrtonyx montezumae*. Avicultural Magazine 118 (2):6-11.

Eitniear, J. C. 2014. Returning the Montezuma Quail to the Eastern Edwards Plateau. Texas Birds Annual 10:28-31.

Leopold, A. S. 1939. Age determination in quail. Journal of Wildlife Management 3:261-265.

Petrides, G. A. and R. B. Nestler 1943. Age determination in juvenal bob-white quail. American Midland Naturalist 30:774-782.

Smith, D. S. and J. R. Cain 1984. Criterion for age classification of juvenile Scaled Quail. Journal of Wildlife Management 48:187-191.

LARK SPARROW BUILDS NEST IN METAL CARPORT

William P. Johnson[1,4], Melanie Hartman,[2] and Erica Thomas[2,3]

[1]US Fish and Wildlife Service, National Wildlife Refuge System, Division of Biological Sciences,
PO Box 277, Canyon, TX 79015, USA.
[2]US Fish and Wildlife Service, Buffalo Lake National Wildlife Refuge, PO Box 179, Umbarger,
TX, 79091, USA.
[3]West Texas A&M University, Department of Life, Earth and Environmental Sciences,
P.O. Box 60808, Canyon, TX 79015, USA.

Lark Sparrows (*Chondestes grammacus*) breed in southern Canada, most of the United States and well into Mexico (Martin and Parrish 2000). They are particularly common in the Great Plains of Texas, Oklahoma, Kansas, and Colorado (Fortin et al. 2005). They occur in a variety of open and semi-open habitats including shrublands, desert scrub, cultivated fields, pastures, prairies and savannah woodlands (Martin and Parrish 2000, Fortin et al. 2005, Holoubek and Jensen 2015).

Lark Sparrows nest on the ground, in shrubs and in small trees (Newman 1970, Bock and Webb 1984, Lusk et al. 2003). Perhaps most common, ground nests are often located near the base of small shrubs or herbaceous plants that offer concealment (Newman 1970, Lusk et al. 2003, Wells and Fuhlendorf 2005). Nests constructed in shrubs and trees tend to be low (average 1.4 to 1.5 m above ground, McNair 1985).

Lark Sparrows occasionally construct nests in those of other passerines, particularly Northern Mockingbirds (*Mimus polyglottos*) (Newman 1970, McNair 1984); these nests appear to serve as a platform, with Lark Sparrows relining the nest or forming a new cup inside it (McNair 1984). Utilization of natural cavities (i.e., cavities formed due to injury or rot) and those formed by woodpeckers seem to be rare (Newman 1970, McNair 1984).

Documentation of nesting in anthropogenic structures is limited to a single nest constructed in the "hollow" of a fence post, with the hollow located at the attachment point of a metal brace (McNair 1984). We report a second observation of Lark Sparrows utilizing a man-made structure for nesting.

Observations were made at Buffalo Lake National Wildlife Refuge, Randall County, Texas. On 17 June 2015, we observed a Lark Sparrow flush from a nest located in an open, four-stall metal carport (34°55'10.94"N, 102°06'41.22"W; Fig. 1); this parking structure is part of the refuge's headquarters complex. The nest was constructed under the roof of the southwest corner of the structure. It was located in the hollow of a structural channel (c-channel) that was affixed atop of a 2.11 m tall (above ground) metal beam (Fig. 2). A 5 cm wide ledge, which supported the nest, was formed by the lower flange of the structural channel. A Lark Sparrow was photographed sitting on the nest on 18 June at 1624 CST (Fig. 2). On 19, 22 and 23 June a Lark Sparrow was observed sitting on the nest. We were unable to observe the nest from 24-29 June. On 30 June no adults were seen at the nest, and the nest appeared to have lost material. All nesting material was gone by 3 July. At no time did we make an attempt to check the nest for eggs or young. The carport was actively used during the period that observations were made.

DISCUSSION

To our knowledge, this is first documentation of a Lark Sparrow nesting in any type of building. Although Lark Sparrows occasionally reuse the nests of other species (Newman 1970, McNair 1984), we do not believe that our observation is a case of nest reuse, as no prior nest was observed in this location of the parking structure. Other species do use the structure for nesting; for instance, Say's Phoebes (*Sayornis saya*) also constructed a nest under the roof of the carport in 2015.

Lark Sparrows are typically associated with ground, shrub and tree nesting (Martin and Parrish 2000), although the percent of the population that nests off the ground (in bushes or trees) may be high in some regions (McNair 1985). Similar to Newman's (1970) report of cavity nesting and McNair's (1985) report of nesting in a fence post, this observation is additional evidence that Lark Sparrow nest sites may be diverse.

---

[4] E-mail: Corresponding Author: bill_johnson@fws.gov

Figure 1. Image of carport located at Buffalo Lake National Wildlife Refuge, Randall County, Texas. The approximate location of the Lark Sparrow nest is indicated by the red arrow. The image was taken while facing north.

Figure 2. Lark Sparrow on nest. Pictures were taken on 19 June 2015, at Buffalo Lake National Wildlife Refuge, Randall County, Texas. Image (A) was photographed from south side of the carport while facing north northwest; image (B) was photographed while inside of parking structure facing west (photographs by M. Hartman).

ACKNOWLEDGMENTS

We are grateful to Mark J. Cancellare for assisting with pictures. Views expressed in this article are those of the authors and do not necessarily represent the opinions of the U.S. Fish and Wildlife Service. Use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

LITERATURE CITED

Bock, C. E. and B. Webb. 1984. Birds as grazing indicator species in southeastern Arizona. Journal of Wildlife Management 48:1045-1049.

Fortin, M. J., T. H. Keitt, B. A. Maurer, M. L. Taper, D. M. Kaufman, and T. M. Blackburn. 2005. Species geographic ranges and distributional limits: pattern analysis and statistical issues. Oikos 108:7-17.

Holoubek, N. S. and W. E. Jensen. 2015. Avian occupancy varies with habitat structure in oak savanna of the south-central United States. Journal of Wildlife Management 79:458-468.

Lusk, J. J., K. S. Wells, F. S. Guthery, and S. D. Fuhlendorf. 2003. Lark Sparrow (*Chondestes grammacus*) nest-site selection and success in a mixed-grass prairie. Auk 120:120–129.

Martin, J. W. and J. R. Parrish. 2000. Lark Sparrow (*Chondestes grammacus*). The Birds of North America. No. 488.

McNair, D. B. 1984. Reuse of other species nests by Lark Sparrows. Southwestern Naturalist 29:506-509.

McNair, D. B. 1985. A comparison of oology and nest record card data in evaluating the reproductive biology of Lark Sparrows, *Chondestes grammacus.* Southwestern Naturalist 30:213-224.

Newman, G. A. 1970. Cowbird parasitism and nesting success of Lark Sparrows in southern Oklahoma. Wilson Bulletin 82:304-309.

Wells, K. M. S. and S. D. Fuhlendorf. 2005. Comparison of microclimate at grassland bird nests with different substrates. Prairie Naturalist 37:21-28.

GREEN PARAKEETS FEEDING ON WOOL-BEARING GALL WASP LARVAE (*ANDRICUS QUERCUSLANIGERA*)

Charles Alexander[1]

270 Fish Hatchery Road, Brownsville, Texas 78520

As noted in Brush (2005), the feeding habits of the Green Parakeet (*Psittacara holochlorus*) have been "poorly known" and are thus little represented in the ornithological literature. While sketching a general diet of "…seeds, fruits, nuts, berries, and probably vegetable matter, procured in the treetops or amongst the outer branches of bushes", Forshaw (1989) also mentions Green Parakeets raiding corn crops in El Salvador and Guatemala. Rodriquez-Estrella et al. (1995) lists a triad of fruits and seeds as the principal diet of P. h. brevipes. Eitniear at al. (1997) describes the crop contents of a *P. h. holochlorus* adult and nestling as containing fruits and dried seeds respectively. A comprehensive study of the ecology of the Green Parakeet across any of its distinct population centers has yet to be undertaken.

Green Parakeets occur throughout Mexico and Northern Central America. Rarely sighted in the Texas Rio Grande Valley's many natural parks and refuges (Brush 2005), they are resident within a number of Valley towns and cities. Large flocks traditionally gather to feed, drink, bathe, and roost-rally on North Tenth Street in McAllen, where they are easily observed.

Since March 2012, I have endeavored to follow and document life history of the Valley's Green Parakeet populations (Alexander 2016). Over the years, my list of Green Parakeet food plants has continued to grow. I have photographed and videoed Green Parakeets feeding on the blossoms of orchid trees (*Bauhinia blakeana*), anacua berries (*Ehretia anacua*), the fruits and flowers of palms (*Washingtonia robusta*), the blossoms and seed pods of crape myrtle (*Lagerstroemia indica)*, hackberries (*Celtis laevigata)*, the acorns of live oak (*Quercus virginiana)*, the fruits and flowers of Mexican olive (*Cordia boissieri*), and the seed pods of Tepehuaje (*Leucaena pulverulenta*). On a regular basis, parakeets have been observed clinging en masse to the sides of buildings while nibbling on clay brick or mortar, a behavior reminiscent of psittacine clay lick geophagy in the neotropics (Brightsmith and Aramburú 2004). For the first eighteen months of my study, I witnessed no evidence of insectivorous feeding among the Valley flocks.

On the afternoon of 30 November 2013, I happened upon a flock of hundreds of Green Parakeets alighting in a live oak tree in an office parking lot along Nolana Street near the Pharr/McAllen city limits. I soon realized that the parakeets weren't feeding on the live oak's plentiful acorns (as they often do), but on fuzzy tan puffballs growing everywhere on the underside of the oak leaves: galls caused by an infestation of the wool-bearing gall wasp (*Andricus quercuslanigera*). The parakeets were clamoring to get at this food source, masticating and shredding each puff to eat the encased larvae within. As I took still photos of the feeding flock, drifts of golden fuzz accumulated on the pavement below, scudding across the parking lot in the wind. Natalie Lindholm, Supervisor of Birds at the Gladys Porter Zoo and my companion for the day's observations, was able to record the flock's activities with my HD video camera.

Drees (2015) has stated that a gall is an " abnormal swelling of plant tissue…caused by mechanical injury or by several species of insects, mites, nematodes, fungi, and bacteria." The common name of *Andricus quercusnigera*—the wool-bearing gall wasp—refers to galls induced on live oak leaves by the life-cycle of these tiny insects. After eggs are deposited on the underside of a host leaf, the tree's immune response envelops the growing wasp larvae in a woolly gall, each round puff containing 2-6 seed-like structures in which the larvae feed and develop. Gall damage to live oaks thus parasitized is considered mainly aesthetic and does not affect the overall health of the tree, though infested trees may drop leaves prematurely (Huber 2014, Buss 2011).

On 28 November 2015, I found a second flock of Green Parakeets feeding on wasp galls, this time

---

[1] E-mail: Karasimbi@yahoo.com

Green Parakeets feeding on wool-bearing gall wasp larvae. Insert- gall on live oak. Photo Charles Alexander.

in a smaller live oak in a parking lot along Tenth Street in north McAllen. As noted in my previous observation, the parakeets were plucking the galls with their bills from the underside of the live oak leaves, manipulating the gall with bill and tongue, some clutching galls in one foot to aid tearing away at the golden fuzz to crunch the wasp larvae encapsulated within. As before, the discarded fuzz floated on the wind and across the parking lot, offering a telltale sign of wasp gall feeding activity. A few yards away, other Green Parakeets, accompanied by a lone Mitred Parakeet (*Psittacara mitratus*) were busy eating the seed pods of crape myrtles.

Though Paula de Faria (2007) states that "parrots are rarely recorded feeding on insects", Forshaw (1989) has postulated "…that parrots as a group are far more insectivorous than is generally suspected." In Australia Yellow-tailed Black Cockatoos (*Calyptorhynchus funereus*) feed extensively in season on the larvae of the longhorn beetle *Tryphocaria acanthocera* and cossid moth *Xyleutes boisduvali*, excavating the heartwood of infested trees in search of insect prey (Barker 1984, Higgins 1999). New Zealand Kaka (*Nestor meridionalis meridionalis*) spend 35% of their feeding time digging *Ochrocydus huttoni* larvae from live mountain beech trunks (Beggs and Wilson 1987). Among New World parrots, Forshaw (1989) lists remains of insects in crop and stomach contents of White-eyed Parakeets (*Psittacara leucophthalmus*). Peach-fronted Parakeets (*Eupsittula aurea*) have been observed consuming Isoptera termites (Sazima 1989, Paula de Faria 2007). In late 19th century Florida, North America's extinct Carolina Parakeet (*Conuropsis carolinensis*) was reported feeding on a millipede outbreak near Lake Istokapoga (Pittsburgh Commercial 1876).

While my McAllen observations constitute the first example of Green Parakeets feeding on Andricus wasp galls, Diaz and Peris (2011) report Austral Parakeets (*Enicognathus ferrugineus*) feeding on *Aditrochus fagicolus* larvae in lenga beech galls. Martuschelli (1994) observed maroon-bellied parakeets (*Pyrrhura frontalis*) feeding on gall-forming Homopteran larvae. In the early 19th century, Carolina Parakeets were observed digging into growths on cottonwood branches, foraging for unspecified larvae (Thomas 1819).

ACKNOWLEDGMENTS

Many thanks to Jack Clinton Eitniear for his constant encouragement and support as I worked to complete this paper, to Natalie Lindholm for her faithful assistance in the field, and to Tim Brush for his long-term interest in the Valley's Green Parakeets.

LITERATURE CITED

Alexander, C. 2016. Flight of the Castaways: Notes from a Wild Parrot Field Journal. Texas Birds Annual 12:33-57.

Barker, R. D. and Vestjens, W. J. M. 1984. The Food of Australian Birds: (I) Non-passerines. Melbourne University Press.

Beggs, J. R. and Wilson, P. R. 1987. Energetics of South Island Kaka (*Nestor meridionalis meridionalis*) Feeding on the Larvae of Kanuka Longhorn Beetles (*Ochrocydus huttoni*). New Zealand Journal of Ecology, Vol 11: 183-187.

Brush, T. 2005. Nesting Birds of a Tropical Frontier: The Lower Rio Grande Valley of Texas. Texas A&M University Press, College Station.

Brightsmith, D. J. and R. Aramburú. 2004. Avian geophagy and soil characteristics in southeastern Peru. Biotropica 36:534–543.

Buss, E. 2011 Facts About Galls On Oaks. http://lee.ifas.ufl.edu/Hort/GardenPubsAZ/facts_about_galls.pdf (accessed November 2013).

Diaz, S. and P., Salvador. March 2011. Consumption of Larvae by the Austral Parakeet (*Enicognathus ferrugineus*). The Wilson Journal of Ornithology 123: 168-171

Drees. 2015. Gall-Making Insects and Mites. Texas A & M Agricultural Extension Service. The Texas A&M University System. http://somervell.agrilife.org/files/2015/03/gall-making-insects.pdf (accessed December 2016).

Eitniear, J. C., Tapia, A. A., and S. M. Mcgehee 1997. Traditional Use of Limestone Cave by Nesting Green Parakeets (*Aratinga holochlora*). Ornitologia Neotropical 8: 243-44.

Forshaw, J. M. Parrots of the World. 1989. 3rd (revised) ed. Lansdowne Editions, Melbourne, Australia.

Higgins, P. J. 1999. Handbook of Australian, New Zealand and Antarctic Birds. Volume 4: Parrots to Dollarbird. Oxford University Press. Melbourne, Australia.

Huber, K. 2014 "Fuzzy galls on oak leaves no cause for alarm". Houston Chronicle, Dec. 8, 2014. http://www.houstonchronicle.com/life/columnists/huber/article/Huber-Fuzzy-galls-on-oak-leaves-no-cause-for-5943625.php *(*accessed December 2016*).*

Martuscelli, P. 1994. Maroon-bellied Conures feeding on gall-forming homopteran larvae. Wilson Bulletin 106:769–770

Paula De Faria, I. 2007. Peach-fronted Parakeet (*Aratinga aurea*) feeding on arboreal termites in the Brazilian Cerrado. Revista Brasileria de Ornitologia 15: 457-458.

Sazima, I. 1989. Peach-fronted Parakeet feeding on winged termites. Wilson Bulletin 10: 656-657.

The Pittsburgh Commercial. Friday, April 28, 1876: 2.

Thomas, D. 1819. Travels through the Western Country in the summer of 1816. David Rumsey, Auburn, New York.

BOOK REVIEWS

FIELD GUIDE TO COMMON TEXAS GRASSES

Hatch, Umphres, and Ardoin, Texas A&M University Press, 2015 Amazon $24.99.

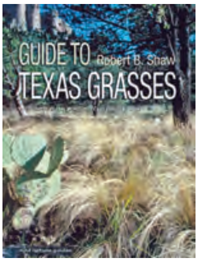
Hatch's **Field Guide to Common Texas Grasses** is a fine and compelling field guide for grasses of Texas. It will be very helpful for birders and plant enthusiasts. It is a very significant improvement on Gould's **Common Texas Grasses**, which it claims specifically to be an update for.

The Introduction is short but jamb-packed with information, and in that respect excellent. It begins by saying it was written for ranchers, conservationists, and life-long learners; by that definition it will include most birders. It is easier reading than, say, Correll and Johnston's **Manual of the Vascular Plants of Texas**, but it is not as polished as one of the favorites for birders – **The Sibley Guide to Birds**, for example. It DOES have a generous, and wonderful, amount of information, but it could benefit from a few editorial improvements and apparent oversights.

The explanations of the abbreviations for Origin, Longevity, and Season located at the end of the Introduction section on page 11 are useful, and these abbreviations are usefully included in the Checklist and in individual Grass Descriptions. As much blank space as this book has, it's odd that the Introduction wasn't a bit more user-friendly and that these and other abbreviations might have been put on a separate page.

The information provided on each species is in depth with one page for each species; that page includes photographs (usually 2) plus line drawings, distribution maps by county, and abbreviations such as N P W to designate Native, Perennial, Warm season, for example. There is also a page for each family briefly stating its characteristics.

Hatch's book can be compared to at least two recent books on grasses



- a large and comprehensive 2012 reference book by Shaw, **Guide to Texas Grasses**

- a relatively small, more intensive 2006 book by Loflin and Loflin, **Grasses of the Texas Hill Country**

Both of these books are also Texas A&M University Press publications. In many ways Hatch's book is a companion book to the comprehensive reference book by Shaw; and the book by the Loflins is a complementary, more detailed book for a specific area, the Hill Country.

172 grasses are covered by Hatch's book, compared to more than 900 in Shaw's book; this may be all that can be done in one field guide, but when you try to use it for a specific region, there may be common grasses that are not covered. For example, we often have widespread, small stands of Texas Tridens in Guadalupe County, but that specific grass is not covered in Hatch (it is covered in Loflin and Loflin).

The distribution maps, in the tradition of other grass and plant books, shows each Texas county and the presence or absence of each grass; this designation apparently depends on actual documented reports or collections by qualified botanists. Often my county, Guadalupe, is shown to be lacking grasses that are clearly present; in some ways the traditional distribution maps for birds in all the common field guides are preferable, showing areas whose outline encompasses where the subject birds have been and can be expected to occur. Of course plants do not have wings, but they do have fairly robust distribution methods.

Grasses of the Texas Hill Country, which is also an excellent book, has only photographs, which are excellent photographs, but it does not have any of the detailed line drawings. It has more species for the Hill Country. In 19 counties it lists 79 species, compared to 172 species in Hatch for 254 counties. However, its definition of "Hill Country" does not agree with other definitions of the hill country or of the Edwards Plateau. The point is this: The Hatch book is very useful state-wide, but if you want more comprehension in a field guide for a specific county or region, you may have to look for additional books.

In summary, the Hatch book is very well done and useful for all levels of plant enthusiasts, including birders looking for help with the identification of grasses they encounter on their many paths through grasslands, savannahs, forests, swamps, and dunes. Birders are used to using multiple books, and they may want multiple books for grasses as well. As in all biological field guides, try to absorb the Introduction with its background, nomenclature, anatomy, and field marks.

Don Schaezler and Charles Tubbs

friends of
BalconesCanyonlands
National Wildlife Refuge

BalconesSongbirdFestival.org

17th Annual Balcones

S♥ngbird Festival

April 21st - 23rd 2017

The only nesting place of the
Golden - cheeked Warbler

Catch sight of the
Black - capped Vireo

Balcones Canyonlands
National Wildlife Refuge
in the Texas Hill Country

Vol. 49. No. 1-2 December 2016 Pages 1-102

CONTENTS

MAJOR ARTICLES

SHORT COMMUNICATIONS

BOOK REVIEW

Slate-colored Redstart *Myioborus miniatus*.

Jack Clinton Eitniear, Editor, E-mail: jclintoneitniear@gmail.com
Kent Rylander, Associate Editor, E-mail : kent.rylander@mac.com

Published in association with Phil Wolfe Graphic Design and Sheridan Press

BULLETIN OF THE TEXAS ORNITHOLOGICAL SOCIETY

Vol. 49 No. 1–2 December 2016